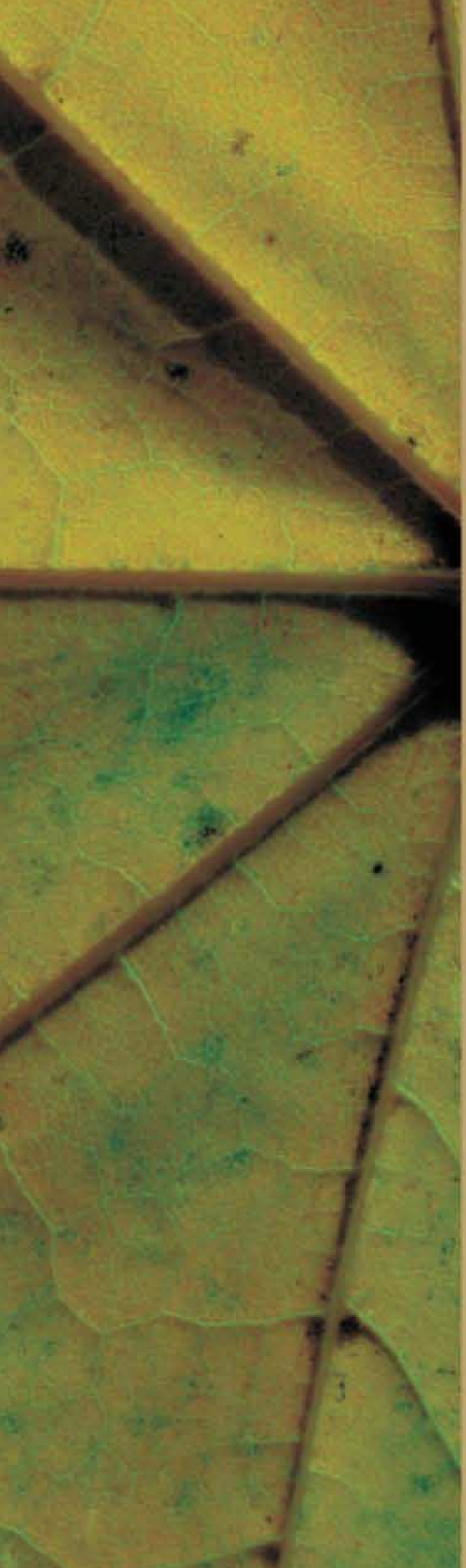


# Acta Silvatica & Lignaria Hungarica

An international Journal in Forest, Wood and Environmental Sciences

Volume 8
2012

ACTA SILVATICA
&
LIGNARIA
HUNGARICA

# Acta Silvatica & Lignaria Hungarica

An International Journal in Forest, Wood and Environmental Sciences

Volume 8
2012

UNIVERSITY OF WEST HUNGARY PRESS

ACTA SILVATICA ET LIGNARIA HUNGARICA
AN INTERNATIONAL JOURNAL IN FOREST, WOOD AND ENVIRONMENTAL SCIENCES
*issued by the Forestry Commission of the Hungarian Academy of Sciences*

*The journal is financially supported by the*
*Hungarian Academy of Sciences (HAS),*
*Faculty of Forestry, University of West Hungary (FF-UWH),*
*Faculty of Wood Sciences, University of West Hungary (FWS-UWH),*
*Hungarian Forest Research Institute (HFRI),*
*Instiute of Wildlife Management, University of West Hungary (IWM-UWH),*
*Sopron Scientists' Society of the Hungarian Academy of Sciences (SSS).*



HU ISSN 1786-691X (Print)
HU ISSN 1787-064X (Online)

*Manuscripts and editorial correspondence should be addressed to*
MAGDOLNA STARK, ASLH EDITORIAL OFFICE
UNIVERSITY OF WEST HUNGARY,
PF. 132, H-9401 SOPRON, HUNGARY

*Phone:* +36 99 518 122
*Fax:* +36 99 329 911
*E-mail:* aslh@nyme.hu
*Information and electronic edition:* http://aslh.nyme.hu

The journal is indexed in the CAB ABSTRACTS database of CAB International; by SCOPUS, Elsevier's Bibliographic Database, by EBSCOhost database and by Versita Sp. z o. o., Warsaw

*Published by* UNIVERSITY OF WEST HUNGARY PRESS,
BAJCSY-ZS. U. 4., H-9400 SOPRON, HUNGARY
*Cover design by* ANDREA KLAUSZ
*Printed by* PALATIA NYOMDA ÉS KIADÓ KFT., SOPRON

The publishing was supported by the joint national-European project TÁMOP 4. 2. 2. B - 10/1 - 2010 - 0018

# Contents

WINKLER, Dániel –TÓTH, Viktória:
Effects of Afforestation with Pines on Collembola Diversity in the Limestone hills of Szárhalom (West Hungary) ..... 9

OLÁH, Viktor – SZŐLLŐSI, Erzsébet – LAKATOS, Ágnes – KANALAS, Péter – NYITRAI, Balázs – MÉSZÁROS, Ilona:
Springtime Leaf Development of Mature Sessile Oak Trees as Based on Multi-Seasonal Monitoring Data ..... 21

KOTROCZÓ, Zsolt – VERES, Zsuzsa – FEKETE, István – PAPP, Mária – TÓTH, János Attila:
Effects of Climate Change on Litter Production in a *Quercetum petraeae-cerris* Forest in Hungary ..... 31

CSÁFORDI, Péter – PŐDÖR, Andrea – BUG, Jan – GRIBOVSZKI, Zoltán:
Soil Erosion Analysis in a Small Forested Catchment Supported by ArcGIS Model Builder ..... 39

LEDIG, F. Thomas:
Climate Change and Conservation ..... 57

BOŽIČ, Gregor – KUTNAR, Lado:
Genetic Variability of two *Fagus sylvatica* (L.) Populations in the South-Western Edge of the Panonnian Plain ..... 75

GÁLOS, Borbála – HÄNSLER, Andreas – KINDERMANN, Georg – RECHID, Diana – SIECK, Kevin – JACOB, Daniela:
The Role of Forests in Mitigating Climate Change – a Case Study for Europe ..... 87

RÉDEI, Károly – CSIHA, Imre – KESERŰ, Zsolt – GÁL, János:
Influence of Regeneration Method on the Yield and Stem Quality of Black Locust (*Robinia pseudoacacia* L.) Stands: a Case Study ..... 103

SZABÓ, Ilona:
Poroid Fungi of Hungary in the Collection of Zoltán Igmándy ..... 113

HORVÁTH, Zsuzsanna – MAROSVÖLGYI, Béla – IDLER, Christine – PECENKA, Ralf – LENZ, Hannes:
Storage Problems of Poplar Chips from Short Rotation Plantations with Special Emphasis on Fungal Development ..... 123

HORVÁTH, Attila László – SZAKÁLOS-MÁTYÁS, Katalin – HORVÁTH, Béla:
Investigation of the Applicability of Multi-Operational Logging Machines in Hardwood Stands ..... 133

TOLVAJ, László – VARGA, Dénes:
Photodegradation of Timber of Three Hardwood Species Caused by Different Light Sources ..... 145

**Technical Reports**

FRANK, Norbert – Heinze, Berthold:
Comparison of Chloroplast Genomes of English Yew (*Taxus baccata* L.) and Japanese Black Pine (*Pinus thunbergii* Parl.) ..... 159

RÉDEI, Károly – KESERŰ, Zsolt:
Target Diameter Models for Leuce Poplar Stands Growing on Sandy Soils ..... 165

**Guide for Authors** ..... 171

**Contents and Abstracts of Bulletin of Forestry Science, issue 2: 1, 2012**
The full papers can be found and downloaded in pdf format from the journal's webpage (www.erdtudkoz.hu) ..... 173

# Tartalomjegyzék

WINKLER Dániel –TÓTH Viktória:
Fenyvesítés hatása a Collembola diverzitásra a Szárhalmi-dombság területén ..... 9

OLÁH Viktor – SZŐLLŐSI Erzsébet – LAKATOS Ágnes – KANALAS Péter – NYITRAI Balázs – MÉSZÁROS Ilona:
Levélnövekedés és a fotokémiai hatékonyság tavaszi felépülése kocsánytalan tölgyfák lombkoronájában, többéves monitoring adatok alapján ..... 21

KOTROCZÓ Zsolt – VERES Zsuzsa – FEKETE István – PAPP Mária – TÓTH János Attila:
A klímaváltozás hatása egy Magyarországi cseres-tölgyes erdő avarprodukciójára ..... 31

CSÁFORDI Péter – PŐDÖR Andrea – BUG, Jan – GRIBOVSZKI Zoltán:
Talajeróziós elemzések egy erdősült kisvízgyűjtőn az ArcGIS Model Builder segítségével ..... 39

LEDIG, F. Thomas:
Klímaváltozás és génmegőrzés ..... 57

BOŽIČ, Gregor – KUTNAR, Lado:
Két bükk populáció genetikai változatossága a Kárpát medence délnyugati peremén ..... 75

GÁLOS Borbála – HÄNSLER, Andreas – KINDERMANN, Georg – RECHID, Diana – SIECK, Kevin – JACOB, Daniela:
Esettanulmány az erdők klímavédelmi szerepének vizsgálatára Európában ..... 87

RÉDEI Károly – CSIHA Imre – KESERŰ Zsolt – GÁL János:
Különböző felújítási módok hatása akácosok hozamára: esettanulmány ..... 103

SZABÓ Ilona:
Magyarország csövestaplói Igmándy Zoltán gyűjteményében ..... 113

HORVÁTH Zsuzsanna – MAROSVÖLGYI Béla – IDLER Christine – PECENKA, Ralf – LENZ, Hannes:
Rövid vágásfordulójú nyár apríték tárolási problémái, különös tekintettel a kifejlődő gombákra ..... 123

HORVÁTH Attila László – SZAKÁLOS-MÁTYÁS Katalin – HORVÁTH Béla:
Többműveletes fakitermelő gépek alkalmazhatóságának vizsgálata magyarországi lombos állományokban ..... 133

TOLVAJ László – VARGA Dénes:

Három lombos fafaj faanyaga fotodegradációs tulajdonságainak összehasonlítása különböző fényforrások alkalmazása esetén ..... 145

**Szakmai beszámolók**

FRANK Norbert – Heinze, Berthold:

Az európai tiszafa és a japán feketefenyő cpDNS összehasonlító vizsgálata ..... 159

RÉDEI Károly – KESERŰ Zsolt:

Homoki Leuce-nyár állományok célátmérő modelljei ..... 165

**Szerzői útmutató** ..... 171

**Erdészettudományi Közlemények 2012. évi számának tartalma és a tudományos cikkek angol nyelvű kivonata**

A tanulmányok teljes terjedelemben letölthetők pdf formátumban a kiadvány honlapjáról (www.erdtudkoz.hu) ..... 173

VERSITA DOI: 10.2478/v10303-012-0001-8 Acta Silv. Lign. Hung., Vol. 8 (2012) 9–20

# Effects of Afforestation with Pines on Collembola Diversity in the Limestone hills of Szárhalom (West Hungary)

Dániel WINKLER[1*] – Viktória TÓTH[2]

[1] Institute of Wildlife Management and Vertebrate Zoology, University of West Hungary, Sopron, Hungary
[2] Institute of Silviculture and Forest Protection, University of West Hungary, Sopron, Hungary

**Abstract** – We investigated the responses of collembolan communities to pine afforestation in an area formerly characterized by a mosaic of autochthonous downy oak woodland and steppe meadows. Study sites were selected in mixed stands of black pine and Scots pine and control samples were taken from downy oak stands and open steppe meadows. A total of 1884 Collembola specimens belonging to 66 species were collected. Three species, namely *Protaphorura pannonica* (Onychiuridae), *Tomocerus mixtus* (Tomoceridae) and *Isotoma caerulea* (Isotomidae) proved to be new to the Hungarian fauna. There are typical Collembola communities which are specific to different habitat types where species of a given composition can only or predominantly be found in that habitat, as well as some basic common species which occur in every habitat. The highest species richness (41) was found in steppe meadows, considerably lower (34) in downy oak forests, reaching the lowest value (25) in pine plantations. Although several forest species present in the oak woodland were completely missing from the pine forests, there was no significant difference between the Collembola diversities of the two forest habitats. The difference became more prominent in collembolan abundance which resulted in less than half of individuals/$m^2$ in pine plantations compared to the soils of downy oak forests, most likely due to the changed soil conditions, especially of humus characteristics, caused by the pine needle litter. Jaccard similarity measure indicated approximately equal similarity (0.24–0.28) for paired comparison, suggesting that a relatively constant 'basic Collembola community' determined by the soil type typical for the area is present; while dissimilarity in communities between sites are partly provided by spatial heterogeneity of open and forest habitats and by the difference of the vegetation type.

**soil fauna / Collembola communities / xerophil habitats / allochthonous pine forest**

**Kivonat – Fenyvesítés hatása a Collembola diverzitásra a Szárhalmi-dombság területén.** Kutatásunkban a fenyvesítés talajlakó ugróvillás-közösségekre gyakorolt hatását vizsgáltuk egy meszes talajú, egykoron molyhos tölgyes sztyepprét élőhelymozaikkal jellemezhető területen. A talajmintákat vegyes erdeifenyves-feketefenyves állományokból, valamint kontrollként őshonos molyhos tölgyesből és sztyepprétről gyűjtöttük. A vizsgálat során összesen 66 faj 1884 egyede került elő. Három faj, a *Protaphorura pannonica* (Onychiuridae), a *Tomocerus mixtus* (Tomoceridae) és az *Isotoma caerulea* (Isotomidae) a hazai faunára nézve újnak bizonyult. A vizsgált élőhelyek jellegzetes ugróvillás-közösségekkel jellemezhetők, amelyek az adott élőhelyre specifikus fajok mellett néhány közös, mindegyik habitatban előforduló fajt is tartalmaznak. A legnagyobb fajszámú (41) Collembola közösséget a sztyeppréteken találtuk, míg a molyhos tölgyesek fajgazdagsága alacsonyabbnak (34) bizonyult. A legkevesebb fajt (25) a telepített fenyvesekben gyűjtöttük. Bár több, a tölgyesekben előkerült erdei faj hiányzott a fenyőállományokból, a két erdei élőhely ugróvillás-közösségének diverzitása között nem mutatkozott szignifikáns eltérés. A közösségek összabundanciáját illetően már

* Corresponding author: dwinkler@emk.nyme.hu; H-9400 SOPRON, Bajcsy-Zs. u. 4.

markánsabb különbségek adódtak: a telepített fenyvesekben a molyhos tölgyeshez viszonyítva kevesebb, mint a fele ugróvillás egyedet mutattunk ki 1 $m^2$-re vetítve. Ez elsősorban a fenyőtűavar lebomlást késleltető hatásának köszönhető, amely ezáltal eltérő humuszformákban is megmutatkozik. A Jaccard-féle fajazonossági index hozzávetőleg azonos (0,24–0,28) értéket mutatott mindegyik élőhelypár esetében. A specifikus eltéréseket az ugróvillás-közösségekben a nyílt és zárt élőhelyek heterogenitása, valamint a vegetációbeli különbségek is okozhatják.

**talajfauna / Collembola közösségek / xerofil habitat / telepített fenyvesek**

## 1 INTRODUCTION

The impact of tree introduction on soil Collembola has been studied less extensively (Arbea – Jordana 1985, Sousa – Da Gama 1994, Sousa et al. 1997, Pinto et al. 1997); although, this kind of conversion process often tends to reduce soil fauna community structure and biodiversity (Elmarsdottir et al. 2008). In Hungary, major land-use changes have been conversion from natural grassland and deciduous woodland to coniferous plantations, using mainly Scots pine (*Pinus sylvestris*), black pine (*Pinus nigra*) and Norway spruce (*Picea abies*). The area covered by such coniferous plantations now represents about 12% of the total forest area in Hungary (Kottek 2008).

The characteristic vegetation types for Fertőmellék Hills are oak forests and steppe meadows. The scattered presence of pine species in the Fertőmellék Hills was already documented from the 1830's, but afforestations with pines of larger tracts of land started only from the 1870's and peaked after World War II (Király 2001). Pines are non-native, the autochthony of Scots pine on schist is questionable (Király 2001). The increasing coverage of pine stands therefore induced considerable changes in landscape, and most presumably, on biodiversity.

The aim of this study was to determine the effects of pine plantations on soil condition and Collembola community composition, species richness, diversity and abundance in a xerophile area. As null hypothesis we considered that there is no difference between the Collembola communities of the pine-forested area and the control forest and meadow areas. As alternative hypothesis we supposed that Collembola community structure, species richness and abundance are different in the pine plantations compared with the autochthonous forest and meadow sites.

## 2 MATERIAL AND METHODS

### 2.1 Study area

The Fertőmellék Hills has a moderately cold – moderately dry climate, with a mean annual air temperature of 9.5–9.8 °C and a mean annual precipitation of 640–660 mm (Dövényi 2010). The study sites are situated between 47°01'39" and 47°02'43" N and 16°44' and 16°45'14" near Fertőrákos, Győr-Moson-Sopron county, Hungary.

Sampling was conducted in mixed allochthonous black pine (*Pinus nigra*) and Scots pine (*Pinus sylvestris*) forests. Autochthonous control sites were selected in both open (steppe meadows) and forest (downy oak stands) habitats.

The main characteristics of the surveyed habitats are given below.

- downy oak stands – closed thermophilous downy oak woodlands. The most dominant canopy species is *Quercus pubescens* with some *Qu. cerris*. The well-developed shrub layer is characterized by *Cornus mas, C. sanguinea, Crataegus monogyna, Ligustrum vulgare, Viburnum lantana. Melica nutans, Convallaria majalis, Hedera helix* and *Campanula trachelium* are frequent in the herb layer. Thermophilous and light-demanding species are sporadic.

- pine plantations – middle-aged mixed stands of black pine and Scots pine, with scattered trees of turkey oak (*Quercus cerris*) and European hornbeam (*Carpinus betulus*) and with a well-developed schrub layer (*Coryllus avellana, Carpinus betulus, Quercus cerris, Cornus mas, C. sanguinea*).
- steppe meadow – *Bromus erectus–Brachypodium pinnatum* xero-mesic grasslands, dry tall herb communities and forest steppe meadows. Most characteristic species are *Brachipodium pinnatum, Festuca rupicola, Carex michelii*. Colonization of the grasslands by shrubs (with *Quercus pubescens, Qu. petraea, Crataegus Monogyna, Prunus spinosa*) have already started.

### 2.2 Soil chemistry

Soil pH was determined by potentiometric methods in $H_2O$ and KCl. Organic carbon, total nitrogen and calcium contents were also assayed. C/N ratio was calculated for each sampling site to find connection between Collembolan abundance and soil conditions.

### 2.3 Sampling design, extraction and taxonomic identification

Sampling was carried out in April 2009. From each habitat, 4 intact soil cores of 100 $cm^3$ were taken with 3 replications from the 0 to 5 cm layer and stored in plastic bags. Collembola were extracted from soil/litter using modified Berlese-Tullgren apparatus at room temperature (Balogh 1958) and preserved in 96% ethyl alcohol until sorting and identification. Species were counted and identified using a binocular microscope according to Gisin (1960), Stach (1960, 1963), Massoud (1967), Deharveng (1982), Fjellberg (1980, 1998), Babenko et al. (1994), Zimdars & Dunger (1994), Weiner (1996), Jordana et al. (1997), Pomorski (1998), Bretfeld (1999), Potapov (2001), Thibaud et al. (2004) and Jordana (2012). Taxonomic classification is primarily based on the most recent classification by Janssens – Christiansen (2011) and on the annotated checklist of the Hungarian Collembola fauna (Dányi – Traser 2008).

### 2.4 Data analysis

In addition to the recorded species richness nonparametric richness estimators (abundance-based estimators *ACE* and *Chao1* and incidence-based estimators *ICE* and *Chao2*) were evaluated using the Species Richness Estimators v2.1 module of www.eco-tools.net. Singletons and doubletons (number of species represented by one or two individuals) were also verified. On species level, the measure 'habitat amplitude' (*HA*) has been used according to the formula of Simpson (1949), which reflects the relative abundance of each Collembola species in the sampled habitats. Rank abundance curves were used to examine general trends in the Collembola dominance structure and abundance for each habitat type. Dominance structure was quantified by using community dominance index (*CDI*), which reflects how large a proportion of the total species present (in terms of numbers of individuals) is made up of the two most abundant species. Two measures of species α diversity were calculated for each habitat: the Shannon index ($H' = -\sum p_i \ln p_i$) and equitability ($J = H' / \ln S$ – where $S$ is species richness). To compare diversity values of two assemblages a *t*-test was used to determine whether they are significantly different (Hutcheson 1970). Rényi diversity profiles (Tóthmérész 1997) were used for partial ranking of the recorded collembolan communities based on diversity. A community of higher diversity has a diversity profile consistently above the profile of a less diverse community. In case the diversity profiles cross each other, the communities are not comparable, and thus the diversity comparison carried out by using *t*-test gets overruled. Community structure comparison between the different habitats was estimated using single linkage cluster analysis based on the *Jaccard* similarity index.

## 3 RESULTS AND DISCUSSIONS

### 3.1 Soil chemistry

The basic physico-chemical parameters of the soil samples are summarized in *Table 1*. Although usually acidification is experienced in the 0–5 cm layer of pine plantations (Halbritter et al. 2007), our pine forest soil samples were close to neutral, and there was no significant differences in soil pH compared with the oak stands or meadow soils (Kruskal-Wallis test). Total calcium content was lower in the soil samples taken from the downy oak forests compared with the pine plantations and steppe meadow area. Total organic carbon content was the highest in the pine forests. The allelochemical inhibition of nitrification by monoterpenes occurring in the pine needle litter is a known fact (Paavolainen et al. 1998). Supposedly, it was one of the reasons why the total nitrogen content proved to be the lowest in this same habitat. The calculated C/N ratio was, hence, the highest in the pine forest soil, indicating a lower organic matter decomposition rate.

*Table 1. Soil physico-chemical parameters of the study sites (mean ± SD)*

| | steppe meadow | downy oak forest | pine plantation | |
|---|---|---|---|---|
| pH $H_2O$ | 7.53 ± 0.12 | 7.37 ± 0.15 | 7.27 ± 0.25 | ns |
| pH KCl | 7.30 ± 0.10 | 7.23 ± 0.21 | 7.16 ± 0.35 | ns |
| Ca (g/kg) | 142.33 ± 10.59 | 95.67 ± 28.36 | 134.42 ± 51.03 | * meadow-oak |
| N (g/kg) | 5.80 ± 1.45 | 6.17 ± 1.99 | 5.77 ± 1.22 | ns |
| C (g/kg) | 104.57 ± 20.32 | 102.23 ± 36.28 | 124.23 ± 19.15 | ns |
| C/N ratio | 18.21 ± 1.48 | 16.51 ± 1.04 | 21.92 ± 4.02 | ns |

** $p < 0.05$; ns – non significant (Kruskal-Wallis test)*

### 3.2 Faunistical results

A total of 1,884 specimens representing 12 families, 38 genera and 66 species of Collembola were collected and identified (*Table 2*), 33 of them are new to the fauna of the Fertőmellék Hills (Traser 2002, Traser et al. 2006). Because of the geographical nature and climate of the area we recorded a number of typical xerothermophilic species (*Xenylla maritima, Doutnacia xerophila, Mesaphorura critica, Metaphorura affinis, Entomobrya handschini, Entomobrya multifasciata, Orchesella albofasciata*).

Three species, namely *Protaphorura pannonica* (Onychiuridae), *Tomocerus mixtus* (Tomoceridae) and *Isotoma caerulea* (Isotomidae) proved to be new to the Hungarian fauna:

*Protaphorura pannonica* (Haybach, 1960)

This Palearctic, euedaphic species typical for meadow habitats, was formerly known from the neighbour countries Austria, Ukraine (Deharveng – Fjellberg 2011) and Romania (Fiera 2007). A total of 5 specimens were found in the samples from the steppe meadow soils.

*Tomocerus mixtus* (Gisin, 1961)

The possible occurrence of this species in the Central European area has already been suggested by Gisin (1961). According to Arbea – Fjellberg (2011), this species is known from Germany (type locality), Austria, Italy, Bosnia and Herzegovina and from the former Yugoslavian countries while it is absent in the major part of Northern, Western and Eastern Europe. We found 7 specimens in the soils samples taken from the pine plantations.

*Isotoma caerulea* Bourlet, 1839

Up to the present time, this species has probably been confounded with *I. anglicana*, a species undoubtedly present throughout the territory of Hungary. The revision of the '*Isotoma viridis* complex' (Fjellberg 2003) made clear the difference between *I. anglicana* and *I. caerulea*, allowing us to identify the latter species for the first time in Hungary taken from the soil samples of the steppe meadows in Szárhalom.

### 3.3 Collembola community analysis

Average abundance and habitat amplitude of the species occurred are presented in *Table 2*, while *Table 3* shows the most important structural characteristics of the Collembola communities found in the three habitats.

*Table 2 Average abundance (individuals/100 cm$^3$) of Collembola species in the habitats studied and species habitat amplitude (HA) according to Simpson's formula*

| COLLEMBOLA | steppe meadow | downy oak forest | pine plantation | *HA* |
|---|---|---|---|---|
| **Neanuridae** | | | | |
| *Deutonura conjuncta* (Stach, 1926) | 0.33 | – | 0.33 | 2.00 |
| *Neanura* sp. | – | – | 0.33 | 1.00 |
| *Pratanurida* sp. | 0.67 | – | – | 1.00 |
| *Pseudachorutes dubius* Krausbauer, 1898 | – | – | 0.33 | 1.00 |
| *Pseudachorutes parvulus* Börner, 1901 | – | 1.00 | 1.67 | 1.88 |
| *Pseudachorutes pratensis* Rusek, 1973 | – | 0.33 | – | 1.00 |
| **Odontellidae** | | | | |
| *Superodontella pseudolamellifera* (Stach, 1949) | – | – | 1.00 | 1.00 |
| **Hypogastruridae** | | | | |
| *Ceratophysella armata* (Nicolet, 1841) | 0.33 | – | – | 1.00 |
| *Ceratophysella luteospina* (Stach, 1920) | 0.33 | 1.67 | 3.00 | 2.10 |
| *Choreutinula inermis* (Tullberg, 1871) | 18.00 | 1.67 | – | 1.18 |
| *Hypogastrura vernalis* (Carl, 1901) | 1.67 | – | – | 1.00 |
| *Schoettella ununguiculata* (Tullberg, 1869) | – | 1.33 | – | 1.00 |
| *Willemia anophthalma* Börner, 1901 | – | 2.00 | – | 1.00 |
| *Xenylla maritima* Tullberg, 1869 | 8.33 | 0.67 | – | 1.16 |
| **Onychiuridae** | | | | |
| *Protaphorura armata* (Tullberg, 1869) | 4.00 | 6.00 | – | 1.92 |
| *Protaphorura cancellata* (Gisin, 1956) | 2.33 | – | – | 1.00 |
| *Protaphorura gisini* (Haybach, 1960) | 5.67 | – | – | 1.00 |
| *Protaphorura pannonica* (Haybach, 1960) | 1.67 | – | – | 1.00 |
| *Protaphorura subarmata* (Gisin, 1957) | 1.33 | – | – | 1.00 |
| *Protaphorura subnemorata* (Gisin, 1957) | – | 16.33 | 7.67 | 1.77 |
| *Protaphorura tricampata* (Gisin, 1956) | – | 2.00 | – | 1.00 |
| **Tullbergiidae** | | | | |
| *Doutnacia xerophila* Rusek, 1974 | 11.67 | 9.00 | 2.00 | 2.32 |
| *Jevania weinerae* Rusek, 1978 | – | 0.33 | 0.33 | 2.00 |
| *Mesaphorura critica* Ellis, 1976 | 4.67 | – | 7.00 | 1.92 |
| *Mesaphorura hylophila* Rusek, 1982 | 2.67 | 1.33 | 11.33 | 1.71 |
| *Mesaphorura italica* (Rusek, 1971) | 1.33 | – | 3.33 | 1.69 |
| *Mesaphorura krausbaueri* Börner, 1901 | 2.67 | – | 5.00 | 1.83 |
| *Mesaphorura macrochaeta* Rusek, 1976 | 2.00 | – | 5.00 | 1.69 |
| *Mesaphorura yosii* (Rusek, 1967) | 0.67 | – | – | 1.00 |
| *Metaphorura affinis* (Börner, 1902) | 1.33 | – | – | 1.00 |
| *Paratullbergia callipygos* (Börner, 1902) | – | 1.33 | – | 1.00 |
| **Oncopoduridae** | | | | |
| *Oncopodura crassicornis* Schoebotham, 1911 | – | 0.67 | 1.67 | 1.69 |
| **Tomoceridae** | | | | |
| *Pogonognathellus flavescens* (Tullberg, 1871) | – | 0.33 | – | 1.00 |
| *Tomocerus mixtus* (Gisin, 1961) | – | – | 2.33 | 1.00 |

*Table 2 cont. Average abundance (individuals/100 cm3) of Collembola species in the habitats studied and species habitat amplitude (HA) according to Simpson's formula*

| COLLEMBOLA | steppe meadow | downy oak forest | pine plantation | *HA* |
|---|---|---|---|---|
| **Isotomidae** | | | | |
| *Anurophorus laricis* Nicolet, 1842 | 0.33 | – | – | 1.00 |
| *Coloburella* cf. *zangherii* (Denis, 1924) | – | 0.67 | 0.67 | 2.00 |
| *Cryptopygus bipunctatus* (Axelson, 1903) | 100.67 | 37.00 | – | 1.65 |
| *Folsomia manolachei* Bagnall, 1939 | 13.33 | 4.00 | – | 1.55 |
| *Folsomia penicula* Bagnall, 1939 | – | 67.00 | – | 1.00 |
| *Folsomia quadrioculata* (Tullberg, 1871) | – | 24.00 | – | 1.00 |
| *Isotoma caerulea* Bourlet, 1839 | 0.33 | – | – | 1.00 |
| *Isotominella minor* (Schäffer, 1896) | 2.67 | 27.00 | 27.00 | 2.19 |
| *Isotomodes productus* (Axelson, 1906) | 0.33 | – | – | 1.00 |
| *Isotomodes sexsetosus* da Gama, 1963 | 0.67 | – | – | 1.00 |
| *Parisotoma notabilis* (Schäffer, 1896) | 1.33 | 23.00 | 30.00 | 2.06 |
| *Proisotoma minuta* (Tullberg, 1871) | – | 1.00 | – | 1.00 |
| **Entomobryidae** | | | | |
| *Entomobrya* sp. juv. | 1.67 | 1.67 | – | 2.00 |
| *Entomobrya handschini* Stach, 1922 | 1.33 | – | – | 1.00 |
| *Entomobrya multifasciata* (Tullberg, 1871) | – | – | 0.33 | 1.00 |
| *Entomobrya muscorum* (Nicolet, 1842) | – | 0.33 | – | 1.00 |
| *Heteromurus nitidus* (Templeton, 1835) | – | 0.33 | – | 1.00 |
| *Lepidocyrtus* cf. *flexicollis* Gisin, 1965 | 0.33 | – | – | 1.00 |
| *Lepidocyrtus lanuginosus* (Gmelin, 1788) | 16.33 | – | – | 1.00 |
| *Lepidocyrtus lignorum* (Fabricius, 1793) | 0.33 | 0.33 | – | 2.00 |
| *Lepidocyrtus paradoxus* Uzel, 1890 | 0.33 | – | – | 1.00 |
| *Lepidocyrtus violaceus* (Geoffroy, 1762) | – | 1.00 | – | 1.00 |
| *Orchesella* sp. juv. | – | – | 0.33 | 1.00 |
| *Orchesella albofasciata* Stach, 1960 | 0.67 | – | – | 1.00 |
| *Orchesella cincta* (Linnaeus, 1758) | – | – | 0.33 | 1.00 |
| *Pseudosinella alba* (Packard, 1873) | 0.33 | 1.67 | – | 1.38 |
| *Pseudosinella* cf. *horaki* Rusek, 1985 | 4.67 | 20.33 | 10.00 | 2.29 |
| *Willowsia nigromaculata* (Lubbock, 1873) | 0.33 | – | – | 1.00 |
| **Neelidae** | | | | |
| *Megalothorax minimus* Willem, 1900 | 0.33 | 13.33 | 8.67 | 1.97 |
| **Katiannidae** | | | | |
| *Sminthurinus elegans* (Firch, 1863) | 7.33 | 1.33 | 0.33 | 1.46 |
| **Sminthuridae** | | | | |
| *Lipothrix lubbocki* (Tullberg, 1822) | – | 0.67 | – | 1.00 |
| *Sminthurus maculatus* Tömösváry, 1883 | 2.00 | – | – | 1.00 |

Only about the 35% of the collected species occurred in more than one habitat, which means that most of the species are habitat specialists. Species with relatively high habitat amplitude were *Doutnacia xerophila*, *Isotominella minor* and *Pseudosinella* cf. *horaki* occurring with high abundance in both meadow and forest habitats.

*Table 3. Estimated species richness and diversity indices of the habitats sampled*

| Habitat | *S* | *N* | *A* | Singletons/ Doubletons | Richness estimates | | Number of presumed species | *H'* | *J* | *CDI* |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | *ACE/ ICE* | *Chao1/ Chao2* | | | | |
| steppe meadow | 41 | 682 | 22733 | 12/4 | 51/81 | 53/55 | 10 – 40 | 2.353 | 0.6336 | 27.86 |
| downy oak forest | 34 | 812 | 27067 | 6/4 | 36/69 | 36/47 | 2 – 35 | 2.517 | 0.7139 | 19.48 |
| pine plantation | 25 | 390 | 13000 | 8/1 | 32/59 | 31/37 | 6 – 35 | 2.446 | 0.7599 | 22.18 |

*S – total number of species; N – number of individuals; A – total abundance (individuals/m$^2$);*
*ACE, ICE, Chao1 and Chao2 – nonparametric richness estimators; H' – Shannon's diversity index;*
*J – equitability; CDI – community dominance index (%)*

The highest overall species richness (41) was found in the steppe meadows (*Figure 1a*), while this parameter decreased in the forest habitats and reached the lowest value (25) in the pine plantation. The highest number of singletons – species represented by single individuals – also occurred in the meadow samples resulting in the highest values of non-parametric species richness estimations *Chao1* and *Chao2* which use the relative proportions of singletons and doubletons for calculating the estimated value. In all three habitats, maximum species richness was predicted by the incidence based estimator *ICE* which takes into account the frequency counts for rare species.

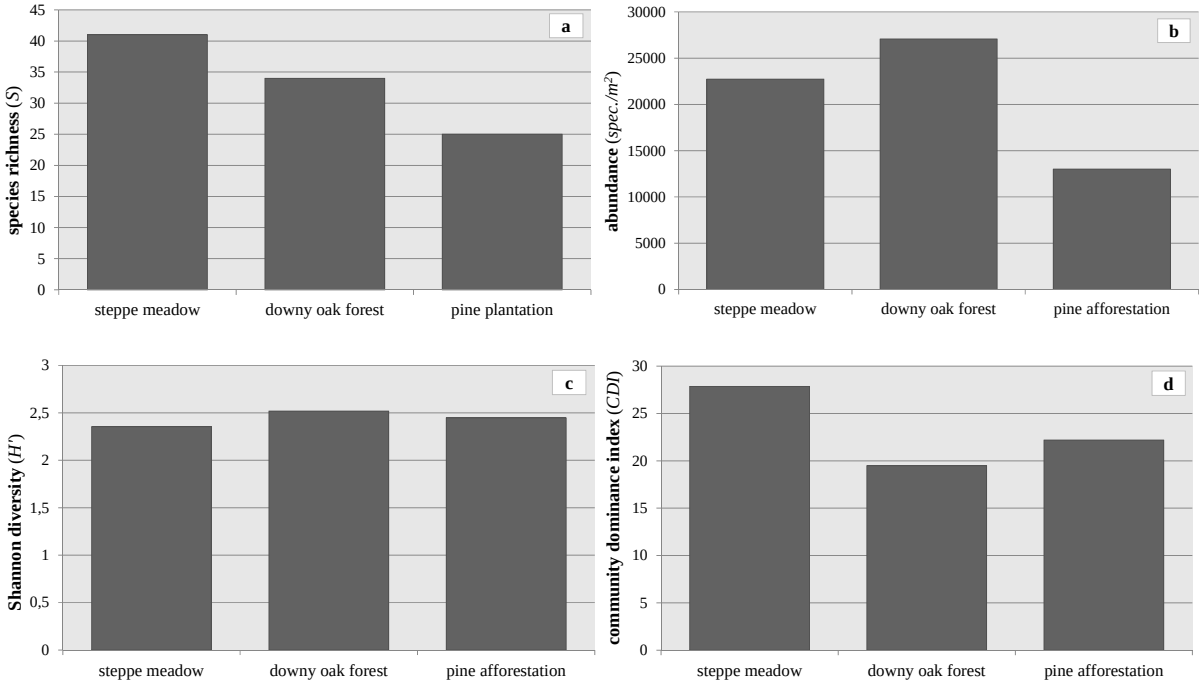



*Fig. 1a-d: Species richness, abundance, Shannon diversity and community dominance index*

The abundance of collembolan communities varied from 13,000 ind./m$^2$ to 27,067 ind./m$^2$ (*Figure 1b*). The most abundant communities were found in the downy oak forests while abundance was less than half in the pine plantations. This phenomenon can probably be explained by changed humus characteristics after the replacement of deciduous oak stands by pine plantations which induced a shift from mull humus towards moder humus

forms. In pine plantations, the decomposition efficiency of the soil-fauna is much lower than in deciduous forests (Dunger – Voigtländer 2009). The decreased organic matter decomposition rate, well predicted by the higher C/N ratio (Lorenz et al. 2004), can therefore be reflected in low Collembola abundance, especially in early stages of pine stands.

Species diversity, measured by Shannon-Weaver index, showed the highest value in the downy oak forests (*Figure 1c*). While diversity profile of the Collembola community found in the pine plantations clearly runs under the downy oak forest's profile (*Figure 2*), Hutheson's modified *t*-test yielded no significant difference between the diversities. The diversity profile of the steppe meadow crosses both forest community profiles meaning that the steppe meadow habitat has a completely different community structure and cannot be ranked with the forest communities based solely on diversity.

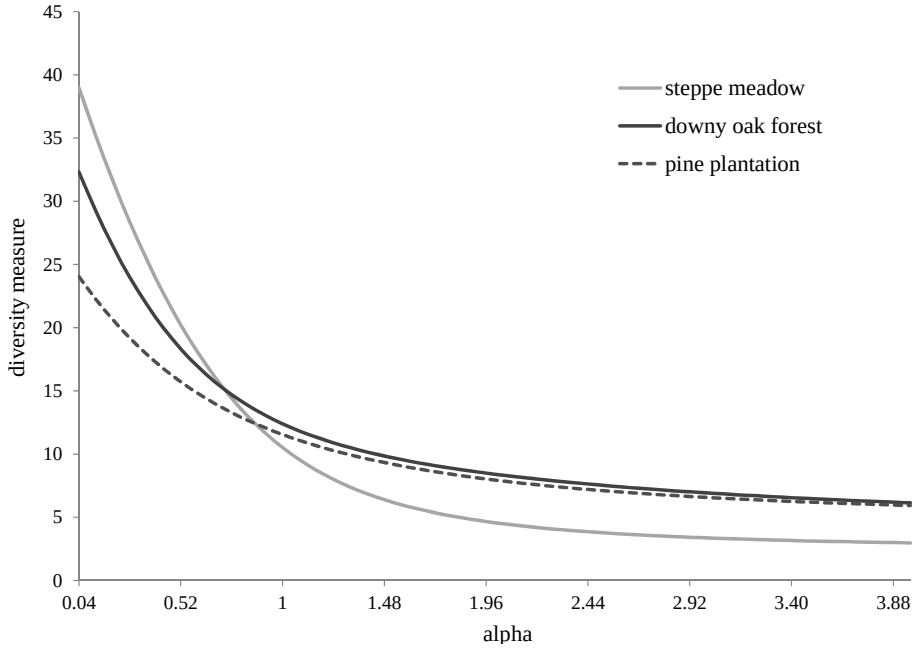



*Figure 2. Diversity profiles of Collembola communities in the different habitats*

Values of the community dominance index (*Figure 1d*) and the species rank abundance curves (*Figure 3*) emphasise well the differences between the dominance structures of Collembola communities found in the sampled habitats.

Dominance index was lowest (~20%) in the downy oak forest which indicates a relatively balanced dominance structure. The most dominant species was *Folsomia penicula*, a rather silvicolous, mesophil species occurring with an average of 67 ind./100 $cm^3$ abundance. Subdominant species were the Palaearctic, xerotolerant *Cryptopygus bipunctatus* and the ubiquist, euedaphic mesophilous *Isotomiella minor*. While most of the *Protaphorura* species are characteristic for open habitats, *P. subnemorata*, recorded also in the downy oak stands, appears to be a typical forest species (Dányi – Traser 2008).

Community dominance structure in the pine plantations shows a similar pattern. The relatively low (~22%) value of *CDI* indicates that there were no species present in extreme abundance. The most dominant species was *Parisotoma notabilis*, accounted for about 12% of the total number of individuals, one of the most ubiquist springtails of the Western Palaearctic, reaching high abundance in both natural and disturbed sites, from forests to open grasslands (Potapov 2001). *Isotomiella minor* appeared in high abundance also in the soil of the coniferous pine forests. While we found a number of common species (*Protaphorura*

*subnemorata, Pseudosinella* cf. *horaki, Isotomiella minor, Parisotoma notabilis*) for both sampled forest types, it is noteworthy to mention that several species (e.g. *Cryptopygus bipunctatus, Folsomia penicula, Folsomia quadrioculata*) occurred in high abundance in the downy oak forests which on the other hand were completely missing in the pine plantations. The very same phenomenon was observed by Traser – Csóka (2001) in their comparative soil fauna study of an autochton pedunculate oak and an allochton pine plantation in South Hungary. Members of the family *Sminthuridae* were also lacking from pine plantation, while they were present in the other two habitats. There were only very few species that were found preferentially in pine plantations (*Pseudachorutes dubius, Superodontella pseudolamellifera, Tomocerina mixtus*).

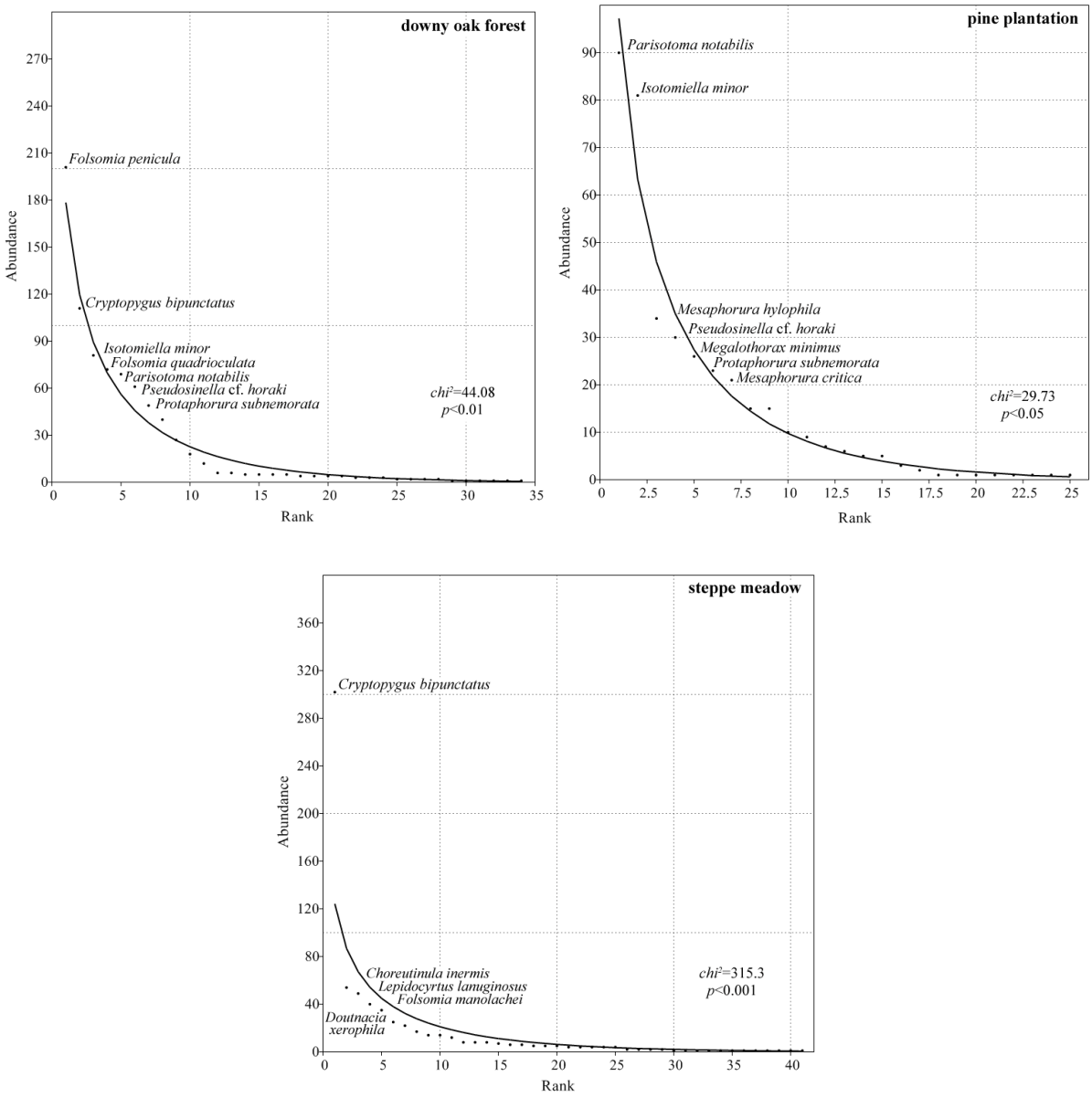



*Figure 3. Rank abundance curves (log series) of Collembola communities of the habitats studied*

The highest dominance index was obtained in the community found in steppe meadows. The dominance structure is therefore somewhat unbalanced and the equitability is lower mainly because of the mass occurrence of the species *Cryptopygus bipunctatus*. The

community is enriched with several rare species represented by few individuals, including typical open area species such as *Hypogastrura vernalis, Protaphorura gisini* or *Metaphorura affinis.*

The agglomerative cluster analysis based on the *Jaccard* measures of similarity (*Figure 4*) shows the separation of meadow and forest habitats. The *Jaccard* similarity values computed for every paired combination are, however, very close to each other (ranging from 0.24–0.28) suggesting that there is a relatively constant 'basic community' as determined by the soil typical type for the area; while dissimilarity in communities between sites is partly provided by spatial heterogeneity of open and forest habitats and by the differences in the vegetation type.

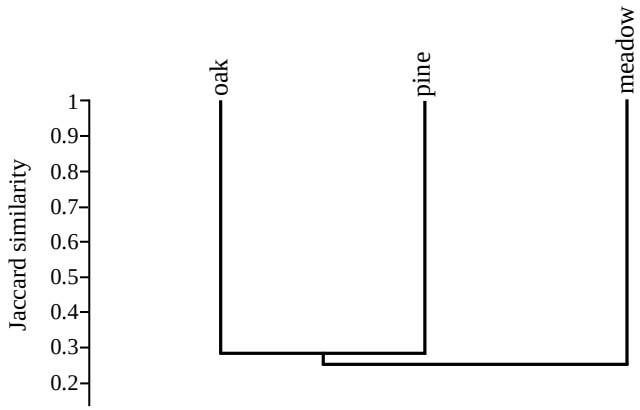



*Figure 4. Dendrogram based on cluster analysis using the Jaccard index of similarity*

## 4 CONCLUSIONS

The study area is characterized by a rich Collembola fauna. There are typical Collembola communities characteristic of the different habitat types studied, which contained specific species that appeared only or predominantly in that habitat, as well as some basic common species occurring in every habitat. Although species richness was lowest in the allochthonous pine forest, species composition still maintained a part of the richness of the autochthonous downy oak forest and even that of the steppe meadow habitat.

Notwithstanding that several forest species present in the oak woodland were completely missing from the pine forests, there was no significant difference between the Collembola diversities of the two forest habitats. The difference became more prominent in collembolan abundance which proved to be less than half of individuals/$m^2$ in pine plantations compared to the downy oak forest soil, and which was probably due to the changed humus characteristics caused by the pine needle litter. The shift towards the moder humus forms, the inhibition of nitrification by monoterpenes present in pine needles can lead to decreased microbial and fungal organic matter decomposition and so thus to a lower Collembola abundance.

Our results suggested that soil conditions have not yet been altered fundamentally that would have been manifested in substantially changed Collembola communities. As a concluding remark it has to be mentioned, that a conversion of pine forests to woodland of native tree species or to steppe meadows has its grounds, and would support the possibility of a quick Collembola community recovery.

**Acknowledgements:** We would like to thank György Nándor Traser for his precious help and comments, Zsófia Varga and Bernadett Varga for their help in soil analysis.

## REFERENCES


ARBEA, J. – FJELLBERG, A. (2011): Tomoceridae. In: DEHARVENG, L. (ed.): Collembola. Fauna Europaea version 2.4, http://www.faunaeur.org, [accession date: 14 August 2012]

ARBEA, J.I. – JORDANA, R. (1985): Efecto de una repoblación con coníferas en un robledal de Navarra sobre los colémbolos edáficos. Bolm. Soc. port. Ent., Supl. 1(2): 277–286.

BABENKO, A.B. – CHERNOVA, N.M. – POTAPOV, M.B. – STEBAEVA, M.B. (1994): Collembola of Russia and adjacent countries: Family Hypogastruridae. Nauka, Moskow. 336 p.

BALOGH, J. (1958): Lebensgemeinschaften der Landtiere. Akadémia kiadó, Budapest-Berlin. 560 p.

BRETFELD, G. (1999): Symphypleona. In: DUNGER, W. ed.: Synopses on Palaearctic Collembola. Vol. 2. Staatliches Museum für Naturkunde, Görlitz. 318 p.

DÁNYI, L. – TRASER, GY. (2008): An annotated checklist of the springtail fauna of Hungary (Hexapoda: Collembola). Opuscula Zoologica 38: 3–82.

DEHARVENG, L. – FJELLBERG, A. (2011): Onychiuridae. In: DEHARVENG, L. (ed.): Collembola. Fauna Europaea version 2.4, http://www.faunaeur.org, [accession date: 14 August 2012]

DEHARVENG, L. (1982): Cle de determination des genres de Neanurinae (Collembola) d'Europe et la region Mediterraneenne, avec description de deux nouveaux genres. - Universite P. Sabatier. Travaux du Laboratoire d'Ecobiologie des Arthropodes Edaphiques 3(4): 7–13.

DÖVÉNYI, Z. (ed.) (2010): Magyarország kistájainak katasztere. MTA Földrajztudományi Kutatóintézet, Budapest. 876 p.

DUNGER, W. – VOIGTLÄNDER, K. (2009): Soil fauna (Lumbricidae, Collembola, Diplopoda and Chilopoda) as indicators of soil eco-subsystem development in post-mining sites of eastern Germany – a review. Soil Organisms 81(1): 1–51.

ELMARSDOTTIR, A. – FJELLBERG, A. – HALLDORSON, G. – INGIMARSDOTTIR, M. – NIELSEN, O.K. NYGAARD, P. – ODDSDOTTIR, E.S. – SIGURDSSON, B.D. (2008): Effects of afforestation on biodiversity. In: HALLDORSSON, G. – ODDSDOTTIR, E.S. – SIGURDSSON, B.D. (eds.): AFFORNORD. Effects of afforestation on ecosystems, landscape and rural development. TemaNord 2008/562: 37–47.

FIERA, C. (2007): Checklist of Romanian springtails (Collembola). Folia Entomologica Hungarica 68: 5–40.

FJELLBERG, A. (1980): Identification keys to Norwegian Collembola. Norsk Entomologisk Forening. 152 p.

FJELLBERG, A. (1998): The Collembola of Fennoscandia and Denmark. Part I.: Poduromorpha. Fauna Entomologica Scandinavica vol. 35: 184 p.

FJELLBERG, A. (2003): Revision of six northern species of the *lsotoma viridis* Bourlet, 1839 complex (Collembola, Isotomidae). Nonv. J. Entomol. 50: 91–98.

GISIN, H. (1960): Collembolenfauna Europas. Museum d'Histoire Naturelle, Genève, 312 p.

GISIN, H. (1961) Collembolen aus der Sammlung C. Börner des Deutschen Entomologischen Institutes. Beiträge zur entomologie 11(3–4): 329–354.

HALBRITTER, A. – CSONTOS, P. – TAMÁS, J. – ANTON, A. (2007): Dolomitsziklagyepek és feketefenyvesek talajainak összehasonlító vizsgálata. In CSONTOS, P. (ed.): Feketefenyveseink kutatása. MTA-ELTE Elméleti Biológiai és Ökológiai Kutatócsoport, Budapest. 15–26.

HUTCHESON, K. (1970): A test for comparing diversities based on the Shannon formula Journal of Theor. Biology 29: 151–154.

JANSSENS, F. – CHRISTIANSEN, K.A. (2011): Class Collembola Lubbock, 1870. In: Zhang, Z.-Q. (Ed.): Animal biodiversity: An outline of higher-level classification and survey of taxonomic richness. Zootaxa 3148: 192–194.

JORDANA, R. – ARBEA, J. I. – CARLOS SIMÓN, M. J. L. (1997): Collembola Poduromorpha. - Fauna Iberica, Vol.: 8. Museo National de Ciencias Naturales, Madrid. 807 p.

JORDANA, R. (2012): Synopses on Palaearctic Collembola: Capbryinae & Entomobryini. Abhandlungen und Berichte des Naturkundemuseums Görlitz 7(1): 390 p.

KIRÁLY, G. (2001): A Fertőmelléki-dombsor vegetációja. Tilia 10: 181–303.

KOTTEK, P. (ed.) (2008): Magyarország erdőállományai, 2006; Adatok. Mezőgazdasági Szakigazgatási Hivatal Központ Erdészeti Igazgatósága, Budapest. http://www.nebih.gov.hu/erdeszet_cd/index.htm [accession date: 14 August 2012]

LORENZ, K. – PRESTON, C.M. – KRUMREI, S. – FEGER, K (2004): Decomposition of needle/leaf litter from Scots pine, black cherry, common oak and European beech at a conurbation forest site. Eur. J. For. Res. 123: 177–188.

MASSOUD, Z. (1967): Monographie des Neanuridae, Collemboles Poduromorphes a piéces buccales modifiées. - Centre National de la Recherche Scientifique, Paris. 399 p.

PAAVOLAINEN, L. – KITUNEN, V. – SMOLANDER, A. (1998): Inhibition of nitrification in forest soil by monoterpenes. Plant and Soil 205: 147–154.

PINTO, C. – SOUSA, J.P. – GRAÇA, M.A. – DA GAMA, M.M. (1997): Forest soil Collembola. Do tree introductions make a difference? Pedobiologia. 41: 207–214.

POMORSKI, J. R. (1998): Onychiurinae of Poland (Collembola: Onychiuridae): Genus, (Supplement), Polish Taxonomical Society, Wroclaw. 201 p.

POTAPOV, M. (2001): Synopses on Palaearctic Collembola: Isotomidae. Abhandlungen und Berichte des Naturkundemuseums Görlitz, 73(2): 603 p.

SIMPSON, E.H. (1949): Measurement of diversity. Nature 163: 688.

SOUSA, J.P. – DA GAMA, M.M. (1994): Rupture in a Collembola community structure from a *Quercus rotundifolia* Lam. forest due to reafforestation with *Eucalyptus globulus* Labill.. European Journal of Soil Biology 30(2): 71–78.

SOUSA, J.P. – VINGADA, J.V. – BARROCAS, H. – DA. GAMA, M.M. (1997): Effects of introduced exotic tree species on Collembola communities: the importance of management techniques. Pedobiologia 41: 166–174.

STACH, J. (1960): The Apterygotan fauna of Poland in relation to the world fauna of this group of insects. Tribe: Orchesellini. Paňstowowe Wydawnictwo Naukowe, Krakowie. 151 p.

STACH, J. (1963): The Apterygotan fauna of Poland in relation to the world fauna of this group of insects. Tribe: Entomobryini. Paňstowowe Wydawnictwo Naukowe, Krakowie. 126 p.

THIBAUD, J.M. – SHULZ, H.J. – DA GAMA, M.M. (2004): Synopses on Palaearctic Collembola: Hypogastruridae. Abhandlungen und Berichte des Naturkundemuseums Görlitz, 75, (2): 287 p.

TÓTHMÉRÉSZ, B. (1997): Diverzitási rendezések. Scientia Kiadó, Budapest 98 p.

TRASER, GY. – CSÓKA, GY. (2001): A mezofauna – Insecta: Collembola – ásotthalmi fenyő- és tölgyerdők talajában. Erdészeti Kutatások 90: 231–240.

TRASER, GY. – SZŰCS, P. – WINKLER, D. (2006): Collembola diversity of moss habitats in the Sopron Region, NW-Hungary. Acta Silv. Lign. Hung. 2: 69–80.

TRASER, GY. (2002): The Collembola of the Fertő–Hanság National Park. In. MAHUNKA, S. ed.: The fauna of the Fertő-Hanság National Park. MTM, Budapest, 259–270.

WEINER, W.M. (1996): Generic revision of Onychiurinae (Collembola: Onychiuridae) with a cladistic analysis. Ann. Soc. Entomol. Fr. (N. S.). 32(2): 163–200.

ZIMDARS, B. – DUNGER, W. (1994): Tullbergiinae. In: DUNGER, W. ed.: Synopses on Palaearctic Collembola. Vol.: I. Abhandlungen und Berichte des Naturkundemuseums Görlitz. Bd. 68/Nr. 3–4. 71 p.

# Springtime Leaf Development of Mature Sessile Oak Trees as Based on Multi-Seasonal Monitoring Data

Viktor OLÁH – Erzsébet SZŐLLŐSI – Ágnes LAKATOS – Péter KANALAS – Balázs NYITRAI – Ilona MÉSZÁROS*

Department of Botany, Faculty of Science and Technology, University of Debrecen, Debrecen, Hungary

**Abstract –** Based on a four year leaf growth data-set we modelled the thermal time-dependent leaf ontogeny in upper and lower canopy layers of mature sessile oak trees, in a *Quercetum petraeae-cerris* forest stand (NE Hungary). Our regression models revealed no considerable differences between the timing of leaf unfolding and leaf expansion of different canopy layers. On the other hand seasonal course in leaf mass-to-area ratio (LMA) indicated that sun leaves needed considerably longer thermal time to fully develop their anatomical structures compared to shade leaves. LMA of sun leaves was higher during the whole leaf maturation process suggesting that 'sun' and 'shade' characteristics develop in very early stage of leaf ontogeny. Functioning of photosynthetic apparatus ($F_v/F_o$) in shade leaves have built up faster and performed better in all developmental stages which could be attributed to two main factors: 1) very early determination of leaf traits as a function of light environment and 2) evolving shading effect of upper canopy layer eliminates photoinhibition in lower leaves.

**leaf ontogeny / leaf traits / leaf morphology / chlorophyll fluorescence / light environment / *Quercus petraea***

**Kivonat – Levélnövekedés és a fotokémiai hatékonyság tavaszi felépülése kocsánytalan tölgyfák lombkoronájában, többéves monitoring adatok alapján.** Jelen munkánk során több levéljellemző változása alapján modelleztük a levélfejlődés folyamatait a hőösszeg függvényében, egy mérsékelt övi cseres-tölgyes erdőállomány idős kocsánytalan tölgyfáinak alsó és felső lombkorona-rétegében. Az eredményeink alapján a levélterület kialakulása a lombkorona alsó és felső részében egyszerre kezdődik és fejeződik be. A fénylevelek esetében a levéltömeg/terület arány (LMA) később érte el a nyári értékét, és a levélfejlődés minden szakaszában nagyobb vastagságot és/vagy nagyobb sűrűséget jelzett, ami a fény- és árnyéklevél-tulajdonságok igen korai kialakulását jelzi. A levelekben a PSII kvantumhozam ($F_v/F_o$) szezonális fejlődését szintén befolyásolja a lombkorona-helyzet. Az árnyéklevelek a teljes fejlődési folyamat alatt magasabb $F_v/F_o$ értékekkel rendelkeztek, és gyorsabban elérték a szezonális maximumukat, mint a fénylevelek, Az utóbbi eredmények szintén a fény- és árnyékjelleg korai determinációjára utalnak, továbbá jelzik azt is, hogy a levélterület-növekedéssel párhuzamosan kialakuló önárnyékolás sietteti a PSII kvantumhozam felépülését.

**levélfejlődés / levéltulajdonságok / levélmorfológia / klorofill fluoreszcencia / fényklíma / *Quercus petraea***

*Corresponding author: immeszaros@unideb.hu; H-4032 DEBRECEN, Egyetem tér 1. Hungary

## 1 INTRODUCTION

Springtime evolution of morphological, anatomical and physiological leaf traits in deciduous communities has fundamental impact on ecosystem functioning (Nizinski – Saugier 1988, Bequet et al. 2011). Leaf area and photosynthetic efficiency as quantitative and qualitative aspects determine carbon assimilating potential of any given plant in the community (Muraoka et al. 2010). Expanding leaves of upper canopy layers affects the functioning of lower layers including certain parts of the plant's own canopy (Umeki et al. 2010, Bequet et al. 2011). In deciduous forests mature trees with multi-layered crown architecture are especially exposed to this shading effect (Valladares – Niinemets 2007, Mészáros et al. 2007, Szőllősi et al. 2010).

A possible strategy for minimizing the effects of self-shading is the proper spatial and temporal distribution of leaves as for instance herbaceous dicots form their foliage (Poorter et al. 2009). Plants applying this strategy of successive leaf phenology form their first leaves at the proximal end of their shoots and the leaf expansion proceeds successively to distal regions of the shoot system thus allowing the younger leaves to capture more light energy (Umeki et al 2010.). Other plant species with nearly simultaneous leaf flushing, such as a number of broad-leaved temperate woody species (Poorter et al. 2009, Szőllősi et al. 2010), optimize their leaf traits in order to gain the best performance (Koike et al. 2001). A typical adaptive mechanism for maximal utilization of the available light energy along the vertical profile of the canopy is formation of sun and shade leaves, respectively (Jurik 1986, Givnish 1988, Yano – Terashima 2001, Terashima et al. 2005, Poorter et al. 2009). Sun leaves are adapted to intense, direct irradiation, higher temperature and higher water vapour deficit by evolving xeromorphic traits (Terashima et al 2005, Mészáros et al. 2007, Poorter et al. 2009). Sun leaves have smaller and thicker leaf lamina, higher leaf mass-to-area ratio (LMA) due to thicker palisade parenchyma or/and larger density in the mesophyll, higher photosynthetic light compensation point and more powerful electron transport system (Yano – Terashima 2001).

Ontogeny of leaves is regulated dynamically in order to properly conform to a given microhabitat, involving e.g. duration of leaf cell division and expansion (Granier – Tardieu 2009). After leaf maturation, physical dimensions of leaves such as leaf area or leaf mass-to-area ratio are irreversibly determined in contrast to e.g. the photosynthetic pigment content, density and formation of chloroplasts (Poorter et al. 2009).

The present study deals with springtime development of leaf traits of mature sessile oak trees with special emphasis on the following questions:

i) How does leaf maturation progress in the case of mature oak trees?
ii) How do various leaf traits develop as a function of thermal time? Is there any difference between lower and upper canopy positions in this respect?
iii) At what phase of leaf maturation do 'sun' and 'shade' characteristics in leaf traits appear?

For studying the above questions we used leaf morphological traits and chlorophyll fluorescence data obtained from multi-seasonal study of mature sessile oak trees (*Quercus petraea* [Matt.] Liebl.).

## 2 MATERIALS AND METHODS

### 2.1 Study site

Leaf development of mature sessile oak trees (*Quercus petraea* [Matt.] Liebl. was monitored in the forest stand of Síkfőkút LTER Europe site (Bükk Mountains, NE Hungary; 47°90'N, 20°46'E, 320–340 m ASL). The site is covered by an approx. 100 years old coppice forest

predominantly consisting of sessile oak (*Quercus petraea*) and Turkey oak (*Quercus cerris*) in the canopy (*Quercetum petraeae-cerris*). The site was established for monitoring the forest structure and functions in 1972 (Jakucs 1985). Meteorological records are collected by a station set up on a 25 m high tower within the forest since 1973. The 30-year average (1973–2002) of annual mean of air temperature and total rainfall was 10.2 °C and 560.5 mm, respectively (*Table 1*). The growing season usually lasts from mid-April to mid-October at the site (Jakucs 1985, Szőllősi et al. 2010).

## 2.2 Meteorological background

In years of this study weather data were recorded automatically in every 30 min on the top of meteorological tower erected above the canopy of forest. Temperature and relative humidity were measured with Hobo ProSeries RH&Temp sensors, rainfall was measured with an automatic rain gauge connected to Hobo Micro Station (both produced by Onset Computer Corporation, Pocasset, USA).

## 2.3 Sampling and measurement

Leaf development was studied in four consequent growth seasons from 2006 to 2009. First leaf samplings were timed shortly after bud-break (mid-April) and were repeated weekly during springtime and later in every second or third week until fully matured state of leaves (late July). Four healthy, approx. 100 years old sessile oak trees were selected for the study (height: 17–20 m, DBH: 28–36 cm) growing close enough to the meteorological tower for easy canopy access.

Leaf samples were harvested on sunny days, between 11:00 a.m. and 15:00 p.m. from the upper (20 m, 'sun leaves') and lower (10 m 'shade leaves') canopy positions of each tree. As a result 10–20 leaves represented each canopy position of each tree.

### *2.3.1 Measured parameter*

Chlorophyll fluorescence: *in vivo* chlorophyll fluorescence measurements were made using PAM-2000 fluorometer (Heinz Walz GmbH, Effeltrich, Germany). Harvested leaves were previously dark-adapted for 30 min in order to fully relax PSII reaction centers and thus gain maximal photochemical efficiency of samples. Minimal ($F_o$) and maximal fluorescence yields ($F_m$) of samples were measured in the middle region of adaxial leaf surfaces avoiding major leaf veins. Variable fluorescence was calculated as $F_v = F_m - F_o$ and was used to characterize the potential quantum yield of PSII photochemistry as the ratio of $F_v/F_o$ (Babani – Lichtenthaler 1996, Lichtenthaler et al. 2005).

Leaf area (LA): a commercial digital scanner was used for leaf area measurements. Scanned images (300 dpi resolution) of leaves were processed with ImageJ image analysis software (Abramoff et al. 2004).

Leaf dry mass (LM): leaves were measured with analytical balance (±0.0001 g accuracy) after drying at 85 °C till constant weight. Based on LA and LM data leaf mass area (LMA = LM/LA, g $dm^{-2}$) of each leaf was calculated.

## 2.4 Data processing

Weather data: as the process of leaf phenology and ontogeny is strongly temperature-dependent (Nizinski – Saugier 1988, Bequet et al. 2011) thermal time – expressed in °C days – was used to describe chronology in development of leaf traits instead of day of year (DOY) basis. Thermal age of leaves was calculated as cumulative daily mean air temperature summed from DOY 100 ($10^{th}$ April) of each year, the date around the leaf unfolding usually begins.

Leaf parameters: respective data of sample trees were collected and merged on sampling date and canopy position basis (20–40 samples per sampling date and canopy position). Medians of leaf traits were used for describing sun and shade canopy layers at a given sampling date. The four-year leaf growth monitoring data base allowed us to utilize 28–28 leaf morphological (LA and LMA) and 27–27 chlorophyll fluorescence ($F_v/F_o$) data points as a total for sun and shade leaves, respectively.

Modeling of leaf maturation: merged leaf trait data series from all sampling years were analyzed as a function of thermal time using non-linear regression function of Sigmaplot 10 (Systat Software, Inc.) in order to mathematically describe leaf expansion. Leaves were considered fully mature from the aspect of a given leaf trait after reaching 95% of their summer value (Cookson et al. 2007) calculated by the regression model. After defining the characteristic phases of leaf maturation, the comparable medians of sun and shade data within a given phase were analyzed by paired Wilcoxon test function of PAST 2.16 (Hammer et al. 2001). Differences were considered to be significant at two-tailed $p < 0.05$ level.

## 3 RESULTS

The annual mean air temperature was higher during the study years as compared to the average of base period from 1973 to 2002 (*Table 1*). In each year the annual mean temperature was above 11.0 °C, and in 2007 it exceeded 12.0 °C. Total rainfall was slightly lower than the 30-year average. Among the study years 2007 was the warmest and driest and 2006 was the most humid in the main period of leaf growth (10th April – 31st July) (*Table 1*).

*Table 1. The annual mean air temperature and total rainfall at Síkfőkút LTER forest site during the study years (2006–2009) compared to the period (1973–2002) and the mean temperature and rainfall in the period of leaf growth (10th April – 31st July)*

| | | 2006 | 2007 | 2008 | 2009 | Average 1973–2002 | Average 2006–2009 |
|---|---|---|---|---|---|---|---|
| **Mean air temperature (°C)** | **Yearly** | 11.3 | 12.2 | 11.4 | 11.3 | 10.2 | 11.6 |
| | **Leaf growth period DOY 100–212** | 18.4 | 19.8 | 18.1 | 18.5 | – | 18.7 |
| **Total rainfall (mm)** | **Yearly** | 588 | 552 | 499 | 554 | 561 | 548 |
| | **Leaf growth period DOY 100–212** | 332 | 168 | 246 | 203 | – | 237 |

### 3.1 Expansion of leaf area

Seasonal development of leaf area showed sigmoid-shape trend with the thermal time (*Figure 1*), that was described by the following function:

$$LA_x = a/(1 + e^{-(x-x_0)/b}) \quad (1)$$

Where:

$LA_x$ is the leaf area at a given thermal time $x$,
$x_0$ is the inflection point of the curve (the point of the maximal LA-growth rate),
$a$ is the summer (maximal) LA,
$b$ is the coefficient of inclination.

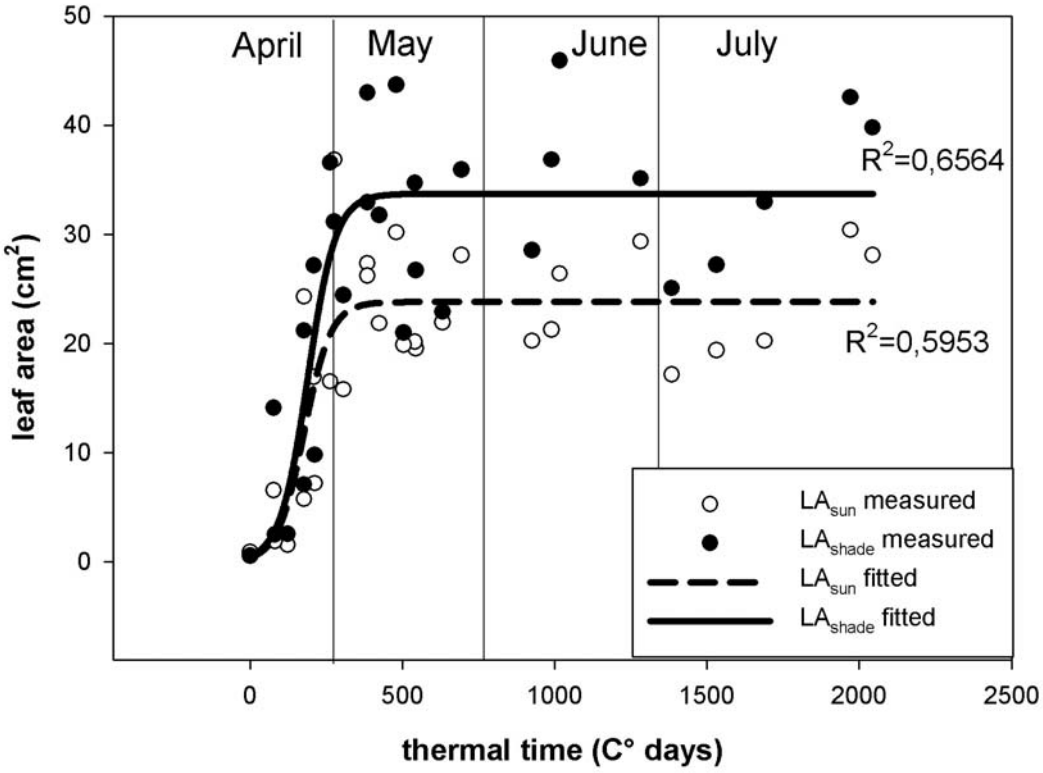



*Figure 1. Temporal course of leaf area (LA) expansion of mature sessile oak trees. Each open and closed circle represents the median of 20–40 leaves collected from the upper (sun) or lower (shade) canopy position of 4 mature sessile oak trees, respectively, at each sampling date characterized by thermal time. Dashed and solid trendlines are calculated seasonal courses of LA in the upper and lower canopy positions, respectively. Vertical lines indicate the average thermal time for the first day of months in the study period 2006–2009.*

LA development begun with a short phase of slower increase ('lag-phase'). After that leaf area expanded rapidly and almost linearly as a function of cumulative temperature until reaching its summer value (Cookson et al. 2007, Bequet et al. 2011) (*Figure 1*). Calculated LA of sun and shade leaves at the start (0 °C days) were similar, that is 0.4 and 0.5 $cm^2$, respectively. Sun and shade leaves reached the summer state of LA nearly at the same thermal age (305 and 323 °C days for sun and shade leaves, respectively). Shade leaves had significantly ($p < 0.001$, $n = 18$) higher summer LA (33.7 $cm^2$) than sun leaves (23.8 $cm^2$), similarly to other reported data on woody species (see e.g.: Koike et al. 2001, Poorter et al. 2009). Assuming that the two types of leaves become mature simultaneously, different rates of leaf expansion could be expected for the two canopy positions (*Figure 1*). Based on regression models, the maximal rates of leaf expansion were 0.136 and 0.185 $cm^2\ cm^{-2}$ °C $days^{-1}$ for sun and shade leaves, respectively. However, the pairwise comparison of medians did not indicate significant difference ($p = 0.105$, $n = 10$).

### 3.2 Seasonal building up of leaf structure

Building up of leaf mass-to-area ratio (LMA) exhibits a seasonal course with three phases (*Figure 2*) (Jurik 1986) and could be described by a peak function:

$$LMA_x = LMA_s + a/(1 + ((x - x_0)/b)^2) \quad (2)$$

Where:

$LMA_x$ is the specific leaf mass area at a given thermal time $x$,
$x_0$ is the thermal time at the point of seasonal minimum,
$LMA_s$ is the summer (maximal) LMA,
$a$ is the constant for the 'amplitude' i.e. seasonal minimal LMA,
$b$ is the constant of curvature.

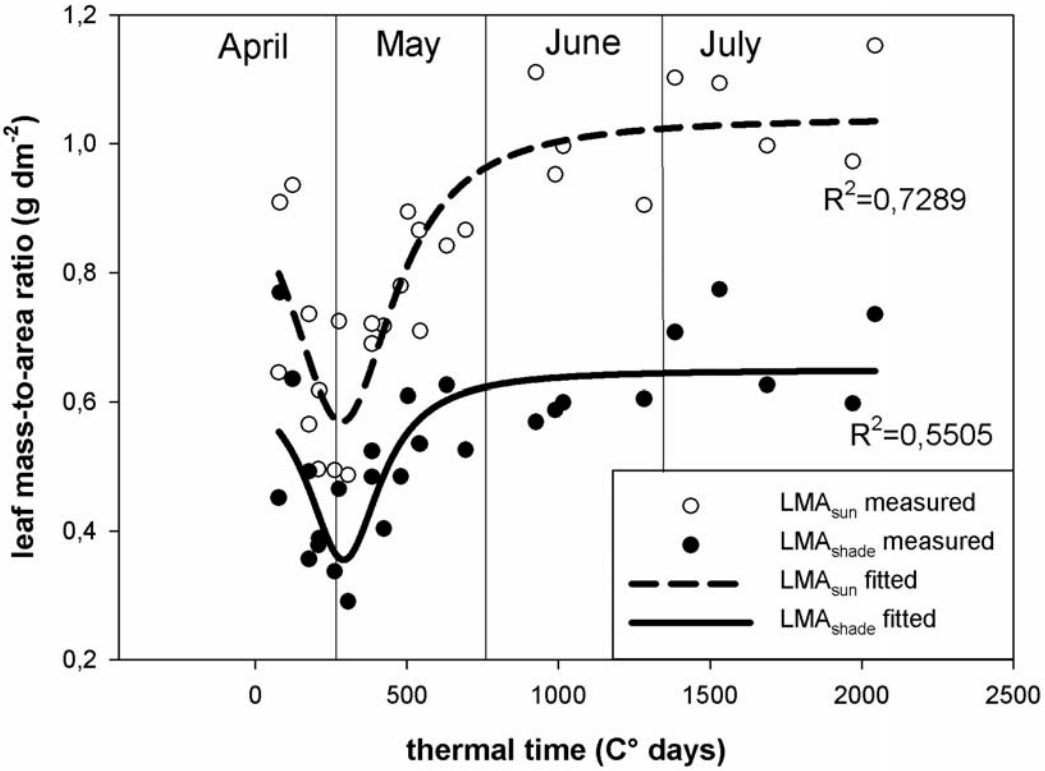



*Figure 2. Temporal course of leaf mass-to-area ratio (LMA) of mature sessile oak trees. Each open and closed circle represents the median of 20–40 leaves collected from the upper (sun) or lower (shade) canopy position of 4 mature sessile oak trees, respectively, at each sampling date characterized by thermal time. Dashed and solid trendlines are calculated seasonal courses of LMA in the upper and lower canopy positions, respectively. Vertical lines indicate the average thermal time for the first day of months in the study period 2006–2009.*

At the start (0 °C days) both sun and shade leaves exhibited relatively high LMA values (0.872 and 0.589 g $dm^{-2}$ for sun and shade leaves, respectively). During leaf expansion LMA fell rapidly till its seasonal minimum in both canopy positions (0.569 and 0.355 g $dm^{-2}$ for sun and shade leaves, respectively) at approximately the same thermal age (284 and 291 °C days for sun and shade leaves, respectively) (*Figure 2*). After that point, LMA increased first rapidly and then moderately until leaves reached their mature state at 864 °C days (sun leaves, LMA = 1.037 g $dm^{-2}$) and 702 °C days (shade leaves, LMA = 0.648 g $dm^{-2}$), respectively (*Figure 2*). The regression models show that LMA of sun leaves was higher in all three developmental stages (0–291 °C days; 291–864 °C days; > 864 °C days) than that of shade leaves, and the pairwise comparisons of LMA medians confirmed these differences (for the three phases $p = 0.004$, $n = 9$; $p = 0.002$, $n = 10$ and $p = 0.004$, $n = 9$, respectively).

### 3.3 Building up of PSII potential quantum yield

Potential quantum yield of PSII photochemistry ($F_v/F_o$) changed during the leaf maturation process and showed a saturating pattern (Babani – Lichtenthaler 1996, Szőllősi et al. 2010) (*Figure 1*) characterized by *eq. (3)*:

$$F_v/F_{0x} = F_v/F_{0\,0} + a \cdot (1 - e^{(-b \cdot x)}) \quad (3)$$

Where:

$F_v/F_{o\,x}$ is the potential photochemical quantum yield at a given thermal time $x$,
$F_v/F_{o\,o}$ is $F_v/F_o$ at thermal time 0 °C day (start),
$F_v/F_{o\,o} + a$ is the summer (maximal) $F_v/F_o$,
$b$ is the coefficient of saturation (curvature).

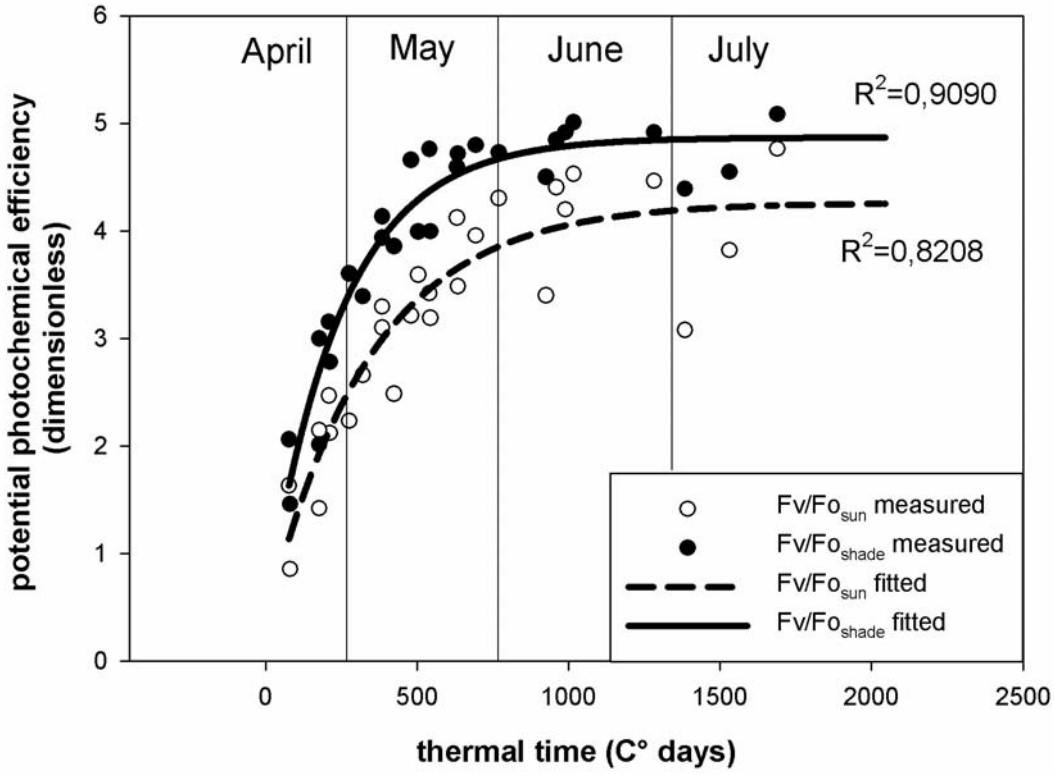



*Figure 3. Temporal course of potential quantum yield of PSII photochemistry ($F_v/F_o$). Each open and closed circle represents median of 20–40 leaves collected from the upper (sun) or lower (shade) canopy position of 4 mature sessile oak trees, respectively, at each sampling date characterized as thermal time. Dashed and solid trendlines are calculated seasonal courses of Fv/Fo in the upper and lower canopy positions, respectively. Vertical lines indicate the average thermal time for the first day of months in the study period 2006–2009.*

$F_v/F_o$ of both sun and shade leaves showed continuous increase after leaf unfolding until reaching their mature state (*Figure 3*). The regression line fitted to $F_v/F_o$ values of shade leaves is running slightly higher than thet of sun leave. At 0 °C days $F_v/F_o$ was 0.45 in shade leaves and 0.34 in sun leaves. The fitted regression curve reflected significantly ($p < 0.001$; $n = 20$) faster development of $F_v/F_o$ in shade leaves (0.018 °C days$^{-1}$ as a maximum) that reached their summer maximum after 724 °C days. $F_v/F_o$ of sun leaves saturated slower

(0.011 °C days$^{-1}$ as a maximum) and required 950 °C days to reach the maximal value. The difference in potential photochemical performances between the two canopy positions did not disappear. $F_v/F_o$ was significantly ($p = 0.015$; $n = 7$) higher in shade leaves (4.867) than in sun leaves (4.261) (*Figure 3*).

## 4 DISCUSSION

Proper timing of leaf unfolding and maturation is essential for deciduous forest trees in order to avoid unfavourable periods and utilize efficiently favourable ones. Leaf phenology of canopy trees and physiological changes during leaf maturation basically determine the functioning of the whole forest ecosystem through their actual carbon balance and – as an additional effect – by shading lower canopy layers.

Based on four-year leaf growth monitoring data set we analysed the leaf maturation process in the upper and lower canopy layers of mature sessile oak trees in a North-east Hungarian forest stand.

As the underlying processes in leaf phenology are strongly temperature-dependent (Nizinski – Saugier 1988, Bequet et al. 2011) we applied thermal age of leaves as the independent variable in our models instead of days of the year. Our models revealed no considerable differences in the timing of leaf unfolding and leaf expansion between the two different canopy layers. LA values at the starting point of our regression models (0 °C day) confirm that both types of leaves unfold at the same time (Poorter et al. 2009, Szőllősi et al. 2010), shortly before DOY 100 (the starting day used in our analysis). From that point leaves required only 3 weeks (approx. 300 °C days) for reaching their fully expanded state, irrespective of their canopy position and of their summer mature LA. Leaves in the lower part of tree crown usually have larger LA than in the upper layers (Koike et al. 2001, Poorter et al. 2009). Our results indicated the similar vertical pattern of LA for sessile oak with 42% higher value for shade leaves. Shade leaves expanded faster than sun leaves and required similar time interval to reach their final LA.

Seasonal course in leaf mass-to-area ratio – which trait mainly refers to the anatomy of leaf mesophyll (Terashima et al. 2005, Poorter et al. 2009) – was different from that of LA (Jurik 1986). As leaf expansion began, LMA dropped sharply indicating the allocation of organic compounds for enlargement of leaf blades and reached its seasonal minimum at around 290 °C days, shortly before the end of leaf expansion (Jurik 1986, Poorter et al. 2009). Sun leaves of sessile oak trees exhibited higher LMA that was comparable with those of other species (Givnish 1988, Koike et al. 2001, Aranda et al. 2004). LMA of sun leaves was higher during the whole leaf maturation process which suggests that 'sun' and 'shade' characteristics develop in the very early stage of leaf maturation (Eschrich et al. 1989, Yano – Terashima 2001, Terashima et al. 2005). The thickening process of leaf blade was different in leaves from upper and lower canopy positions. The former ones required considerably longer thermal time for the full development of anatomical structures than the latter ones.

Characteristics of leaf development in dicotyledonous plants differ from that of monocotyledonous species (Croxdale and Omasa 1990, Granier – Tardieu 2009). In contrast to monocotyledonous species where assimilating tissues reach their mature state right after leaving the growing zone of leaf, the dicotyledonous leaves undergo gradual maturation, which is not uniform across the whole leaf blade but progresses from the leaf tip to the leaf basis. This maturation process is reflected by the seasonal course of potential quantum yield of PSII photochemistry ($F_v/F_o$) which shows a saturating pattern (Babani – Lichtenthaler 1996). Similar to LMA, photosynthetic apparatus of shade leaves matured faster and performed better in all developmental periods as compared to sun leaves. This difference

could be attributed to two factors: 1) very early determination of leaf traits as a function of light environment (Yano – Terashima 2001), which results in anatomical specialization (Eschrich et al. 1989) allowing a faster building up of summer $F_v/F_o$ in shade leaves, and 2) the chronic photoinhibition of sun leaves (Babani – Lichtenthaler 1996, Lichtenthaler et al. 2005). At the beginning of leaf development, both the structure of photosynthetic apparatus and defence mechanisms against excess light are far from their maximal efficiency resulting in considerable photoinhibition of photosystems (Croxdale – Omasa 1990, Szőllősi et al. 2010). When the tree canopy is approaching its summer state – from the aspect of LA and LMA – photoinhibition gradually decreases in leaves of lower canopy layers as a result of shading effect by upper layers. Sun leaves as an opposite have to cope with photoinhibition during their entire lifespan which results in lower (Lichtenthaler et al. 2005) and slower development of $F_v/F_o$ due to the significant investment into defence against excess light.

**Acknowledgements:** Funding was provided by National Research Foundation Projects (OTKA No. K68397), TÁMOP-4.2.2/B-10/1-2010-0024 and Life08 ENV/IT/000399 Project. The authors are grateful to Katalin Szabó for technical assistance in measurement of leaf traits.

## REFERENCES

ABRAMOFF, M.D. – MAGELHAES, P.J. – RAM, S.J. (2004): Image Processing with ImageJ. Biophotonics International 11 (7): 36–42.

ARANDA, I. – PARDO, F. – GIL, L. – PARDOS, J. (2004) Anatomical basis of the change in leaf mass per area and nitrogen investment with relative irradiance within the canopy of eight temperate tree species. Acta Oecol. 25 (3): 187–19.

BABANI, F. – LICHTENTHALER, H.K. (1996): Light-induced and age-dependent development of chloroplasts in etiolated barley leaves as visualized by determination of photosynthetic pigments, $CO_2$ assimilation rates and different kinds of chlorophyll fluorescence ratios. J. Plant Physiol. 148 (5): 555–566.

BEQUET, R. – CAMPIOLI, M. – KINT, V. – VANSTEENKISTE, D. – MUYS, B. – CEULEMANS, R. (2011): Leaf area index development in temperate oak and beech forests is driven by stand characteristics and weather conditions. Trees 25 (5): 935–948.

COOKSON, S.J. – CHENU, K. – GRANIER, C. (2007): Day length affects the dynamics of leaf expansion and cellular development in *Arabidopsis thaliana* partially through floral transition timing. Ann. Bot. 99 (4): 703–711.

CROXDALE, J.G. – OMASA, K. (1990): Patterns of chlorophyll fluorescence kinetics in relation to growth and expansion in cucumber leaves. Plan. Physiol. 93 (3): 1083–1088.

ESCHRICH, W. – BURCHARDT, R. – ESSIAMAH, S. (1989): The induction of sun and shade leaves of the European beech (*Fagus sylvatica* L.): anatomical studies. Trees 3 (1): 1–10.

GIVNISH, T.J. (1988): Adaptation to sun and shade: A Whole-plant Perspective. Aust. J. Plant Physiol. 15 (2): 63–92.

GRANIER, C. – TARDIEU, F. (2009): Multi-scale phenotyping of leaf expansion in response to environmental changes: the whole is more than the sum of parts. Plant, Cell Environ. 32 (9): 1175–1184.

HAMMER, Ø. – HARPER, D.A.T. – RYAN, P.D. (2001): PAST: paleontological statistics software package for education and data analysis. Palaeontologia Electronica. Vol. 4 (No): 1. Available at: http://palaeo-electronica.org/2001_1/past/issue1_01.htm.

JAKUCS, P. (1985): Ecology of an oak forest in Hungary. Akadémiai Kiadó, Budapest, pp. 19–32.

JURIK, T.W. (1986): Temporal and spatial patterns of specific leaf weight in successional northern hardwood tree species. Am. J. Bot. 73 (8): 1083–1092.

KOIKE, T. – KITAO, M. – MARUYAMA, Y. – MORI, S. – LEI, T.T. (2001): Leaf morphology and photosynthetic adjustments among deciduous broad-leaved trees within the vertical canopy profile. Tree Physiol. 21 (12–13): 951–958.

MÉSZÁROS, I. – VERES, S. – KANALAS, P. – OLÁH, V. – SZŐLLŐSI, E. – SÁRVÁRI, É. – LÉVAI, L. – LAKATOS, Gy. (2007): Leaf growth and photosynthetic performance of two co-existing oak species in contrasting growing seasons. Acta Silvatica et Lignaria Hungarica 3: 7–20.

MURAOKA, H. – SAIGUSA, N. – NASAHARA, N.K. – NODA, H., YOSHINO, J. – TAKU, M.S. – NAGAI, S. – Murayama, S. – Koizumi, H. (2010): Effects of seasonal and interannual variations in leaf photosynthesis and canopy leaf area index on gross primary production of a cool-temperate deciduous broadleaf forest in Takayama. J. Plant Res. 123 (4): 563–576.

LICHTENTHALER, H.K. – BUSCHMANN, C. – KNAPP, M. (2005): How to correctly determine the different chlorophyll fluorescence parameters and the chlorophyll fluorescence decrease ratio RFd of leaves with the PAM fluorometer. Photosynthetica. 43 (3): 379–393.

NIZINSKI, J.J. – SAUGIER, B. (1988): a model of leaf budding and development for a mature *Quercus* forest. J. Appl. Ecol. 25 (2): 643–652.

POORTER, H. – NIINEMETS, U. – POORTER, L. – WRIGHT, I.J. – VILLAR, R. (2009): Causes and consequences of variation in leaf mass per area (LMA): a meta-analysis. New Phytol. 182 (3): 565–88.

SZŐLLŐSI, E. – OLÁH, V. – KANALAS, P. – KIS, J. – FENYVESI, A. – MÉSZÁROS, I. (2010): **S**easonal variation of leaf ecophysiological traits within the canopy of *Quercus petraea* (Matt.) Liebl. trees. Acta Biologica Hungarica 61 (Suppl): 172–88.

TERASHIMA, I. – ARAYA, T. – MIYAZAWA, S.I. – SONE, K. – YANO, S. (2005): Construction and maintenance of the optimal photosynthetic systems of the leaf, herbaceous plant and tree: An eco-developmental treatise. Ann. Bot. 95 (3): 507–519.

UMEKI, K. – KIKUZAWA, K. – STERCK, F.J. (2010): Influence of foliar phenology and shoot inclination on annual photosynthetic gain in individual beech saplings: A functional–structural modeling approach. Forest Ecol. Manag. 259 (11): 2141–2150.

VALLADARES, F. – NIINEMETS, Ü. (2007): The architecture of plant crowns: from design rules to light capture and performance. In: PUGNAIRE, F.I. – VALLADARES, F. (eds.): Handbook of functional plant ecology. CRC, Boca Raton, pp. 101–149.

YANO, S. – TERASHIMA, I. (2001): Separate localization of light signal perception for sun or shade type chloroplast and palisade tissue differentiation in *Chenopodium album*. Plant Cell Physiol. 42 (12): 1303–1310.

VERSITA DOI: 10.2478/v10303-012-0003-6 Acta Silv. Lign. Hung., Vol. 8 (2012) 31–38

# Effects of Climate Change on Litter Production in a *Quercetum petraeae-cerris* Forest in Hungary

Zsolt KOTROCZÓ[a *] – Zsuzsa VERES[a] – István FEKETE[c] – Mária PAPP[b] – János Attila TÓTH[a]

[a] Department of Ecology, University of Debrecen, Debrecen, Hungary
[b] Department of Botany, University of Debrecen, Debrecen, Hungary
[c] Institute of Environmental Science, Nyíregyháza College, Hungary

**Abstract – C**limate change is a global problem. During the last century the increase of annual average temperature was 0.68°C, while the decrease of annual average of precipitation was 83 mm in Hungary. According to the long term meteorological data of Síkfőkút forest ILTER site the annual average temperature increased while average of yearly precipitation decreased, the forest climate became warmer and dryer. These processes could considerably contribute to forest decline, not only in the *Quercetum petraeae-cerris* stand of Síkfőkút, but everywhere in the country. Species composition and structure of the forest have changed considerably, as 68% of sessile oak (*Quercus petraea*) and 16% of Turkey oak (*Quercus cerris*) have died. Forest decline resulted in the breaking up of the formerly closed canopy, and consequently, in the formation of gaps in the forest. In the gaps, a secondary canopy developed with tree species of less forestry value. As a consequence, mass regeneration of field maple (*Acer campestre*) appeard in the gaps. The formation of gaps accelerated the warming and aridity of forests. In the article we answer the following question: how did climatic change and changing forest structure influence the leaf-litter production in the last four decades?

**oak forest / oak decline / Síkfokút Project / litter production / climate change**

**Kivonat – A klímaváltozás hatása egy Magyarországi cseres-tölgyes erdő avarprodukciójára.** Napjainkban a klímaváltozás egy globális probléma. Magyarországon az elmúlt évszázad folyamán az évi átlagos hőmérséklet 0,68 °C-ot emelkedett, az évi csapadék átlag 83 mm-t csökkent. A Síkfőkút Project korábbi, közel 20 éves meteorológiai adatsorai is bizonyítják a klímaváltozást, az erdő klímája melegebbé és szárazabbá vált. Mindezen folyamatok is hozzájárulhattak ahhoz a fapusztulási folyamatokhoz, amely nem csak a Síkfőkúti tölgyes állományt, de az ország számos állományát érintette. Az erdő fafaj összetétele és struktúrája jelentős mértékben megváltozott, a kocsánytalan tölgy 68%-a, a csertölgy 16%-a kipusztult. A fapusztulás következtében a korábbi zárt lombkorona felnyílt, az elpusztult fák helyén kisebb nagyobb lékek alakultak ki, amelyekben erdészeti szempontból kevésbé értékes fafajok nőttek fel és hozták létre a második lombkoronaszintet. Ennek köszönhető a mezei juhar tömeges megjelenése és térhódítása a lombkoronaszintben. A lékek kialakulása, a klímaváltozás hatását fokozva, felgyorsította az erdő felmelegedését és szárazodását. Jelen dolgozatunkban arra kerestük a választ, hogy a klímaváltozás, és ezen keresztül az erdő struktúrájának a változása hogyan befolyásolta az avarprodukciót az elmúlt négy évtized folyamán.

**tölgyerdő / tölgypusztulás / Síkfőkút Project / avarprodukció / klímaváltozás**

* Corresponding author: kotroczo.zsolt@gmail.com; H-4032 DEBRECEN, Egyetem tér 1.

## 1 INTRODUCTION

Litter fall exerts a great influence on physical, chemical and biological characteristics of soil (Pande et al. 2002). Litter production in forest ecosystems is determined by climatic condition, species composition and successional stage (Haase 1999). Factors like climate, season, canopy architecture, timing of canopy gap formation and insect outbreaks govern the dynamics of the forests in terms of its structure and function of this ecosystem (Sukumar et al. 1992).

In Hungary, the annual average temperature increase was 0.68 °C, while the annual average of precipitation decrease was 83 mm during the last century (Szalai 2004, Jolánkai et al. 2004). During the last few decades the long-term meteorological data of the Síkfőkút Forest have followed the global climate change, the forest climate became dryer and warmer. Due to the climate change the forest structure and species composition have been changing dramatically. Since 1972 68% of *Quercus petraea* and 16% of *Q. cerris* trees died and in the gaps new tree species (*Acer campestre, A. tataricum, Cerasus avium, Carpinus betulus*) have grown up from the shrub layer into the canopy layer (Bowden et al. 2006, Kotroczó et al. 2007). The new species created a second canopy layer. Forest decline has been reported since the beginning of the 1970s in Central Europe as well as in North America (Siccama et al. 1982, Klein – Perkins 1987, Bruck – Robarge 1988). Subsequently, the extension of mortality became a chronic problem in the European oak forests (Gaertig et al. 2002, Thomas et al. 2002, Mátyás 2004, Helama et al. 2009, Mátyás 2010). Extensive decay of sessile oak has been reported in numerous European countries (Brasier et al. 1993, Chappelka – Freer-Smith 1995, Thomas – Büttner 1998).

At the Síkfőkút site (Hungary), decline affected both forest stand components (sessile oak – *Quercus petraea* and Turkey oak – *Q. cerris*) however, it affected sessile oak considerably stronger. In case of Turkey oak, higher vitality can be explained by its better adaptability to warming and drying climate. The adaptability is presumably connected to the higher water storage capacity which was pointed out inside the trunk of this submediterranean oak species (Béres et al. 1998). As compared to the survey which was carried out in 1973 (Jakucs 1985), no regeneration of oak appeared on the study area during the last few decades, although it may be suspeced that game damage contributed to this situation. Mass regeneration and spreading of common maple characterizes the canopy level. Gaps in the canopy level accelerated the warming and aridification of the forest.

How the litter production is influenced by climate change, is still relatively unknown. Some authors (Tóth et al. 2007) predicted that although warming may increase litter production, aridity reduces litter production, and the combined effects of the two processes will determine the result of changes and the direction of the trend. Other authors explained these changes by shifts in species composition due to oak decay (Kotroczó et al. 2007). The litter production changes can also affect climate change due to drought-induced forest boundary shift towards the poles (Mátyás 2010). Clearly, litter production is controlled by climate as well as edaphic factors (Meentemeyer et al. 1982).

This study was aimed at quantifying litter production of a *Quercetum petraeae-cerris* forest. Considering the local environmental factors, the aim of our research was to examine the extent of changes in leaf litter production, under the effect of modifications in the tree species composition of the forest comparing the surveys of 1972–1976 and of 2003–2010.

## 2 METHODS

### 2.1 Area and project descriptions

Our research was carried out at the Síkfőkút site established in 1972 by Jakucs (1985) for the long-term study of forest ecosystems. The experimental site of 27 ha is located in the south part of the Bükk Mountains in North Eastern Hungary at 325 m altitude. GPS coordinates are N 47°55' E 20°28'. The forest has been protected since 1976, and it is part of the Bükk National Park at present. Annual precipitation amounts to 550 mm. According to the FAO Soil Classification, the type of soil is Cambisos (Fekete et al. 2011). The forest is a semi-natural stand (*Quercetum petraeae-cerris* community) without forest management. The Síkfőkút Project is the member of the Hungarian LTER (Long Term Ecolgical Research) and the ILTER (International Long Term Ecological Research) networks since 1995 (Kovács-Láng et al. 2000).

### 2.2 Measuring of litter production

The litter production of Síkfőkút Forest was continuously measured from 1972 to 1976 (Tóth et al. 1985). From 2003 to present we renewed these measurements to obtain information about the effect of climate change on the litter production. Litter inputs were measured by 30 plastic boxes of 55.5 × 36.5 cm (0.2 $m^2$) size. The total surface of these boxes is 6 $m^2$. The plastic boxes were placed randomly at the research site. The litter production was collected monthly and separated into the following parts: leaf litter (every species separately), branch litter and other debris. Fresh litter weights were converted to oven-dry weights (85 °C for 48 hours).

We used one-way ANOVA with the Tukey's HSD test as a post-hoc test if necessary to determine the significant differences among litter production of tree species. All statistical analysis were performed using PAST statistical software.

## 3 RESULTS AND DISCUSSION

### 3.1 Long-term changes of the litter production

Comparison of the earlier period (between 1972 and 1976) and the present term (between 2003 and 2010) showed that leaf-litter production of *Quercus petraea* is half of that measured during the former survey (*Figure 1*), and significantly higher between 1972 and 1976 than between 2003 and 2010 ($F = 84.75$, $p < 0.001$). This observation can be explained by the considerable mortality (68%) of sessile oak observed during the past forty years.

Despite of the 16% mortality of *Quercus cerris*, its leaf-litter production did not decrease in comparison with the survey between 1972 and 1976. Vigorous growth of *Quercus cerris* not only compensated but significantly surpassed its former leaf-litter production in years 2003–2010 ($F = 11.29$, $p = 0.006$) (*Figure 2*).

Leaf-litter production of *Acer campestre* is nearly five times as much as it was in the former period, and differences were significant ($F = 18.06$, $p = 0.001$).This can be explained by the fact that *Acer campestre* formerly occurred in the shrub layer, while presently it forms a part of canopy-layer, where it has become the second most common tree species, representing 28.17% of the total number of trees (*Figure 3*).

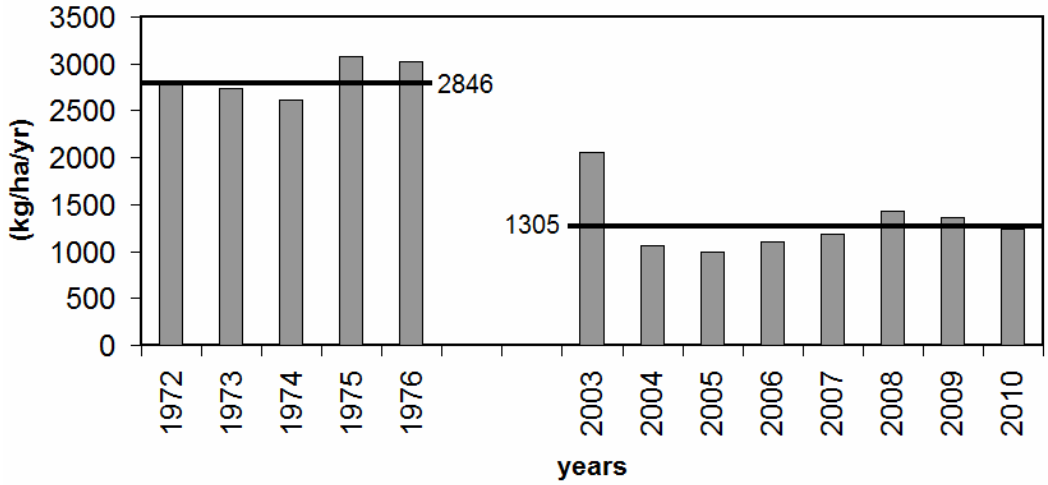



*Figure 1. Annual litter production of Quercus petraea (with periodic averages) between 1972 and 2010 (kg/ha/yr)*

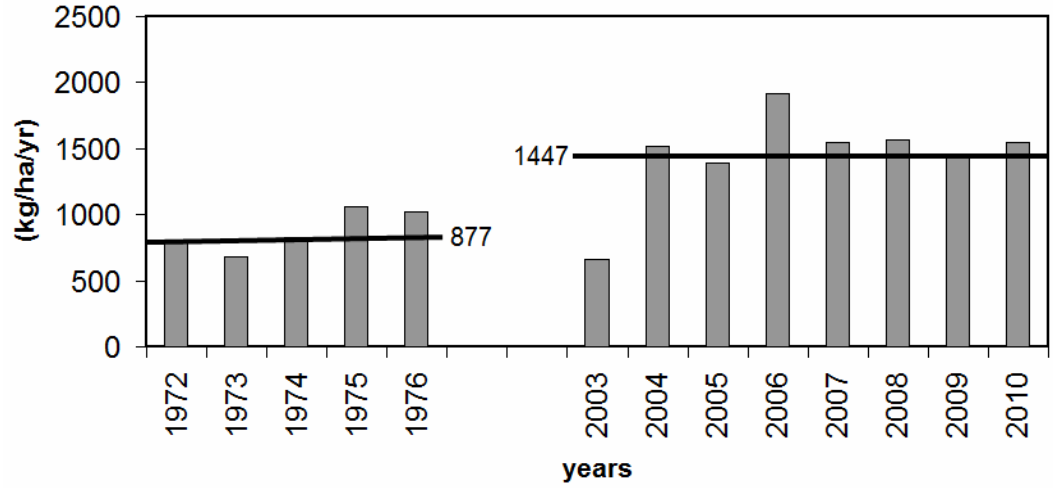



*Figure 2. Annual litter production of Quercus cerris (with periodic averages) between 1972 and 2010 (kg/ha/yr)*

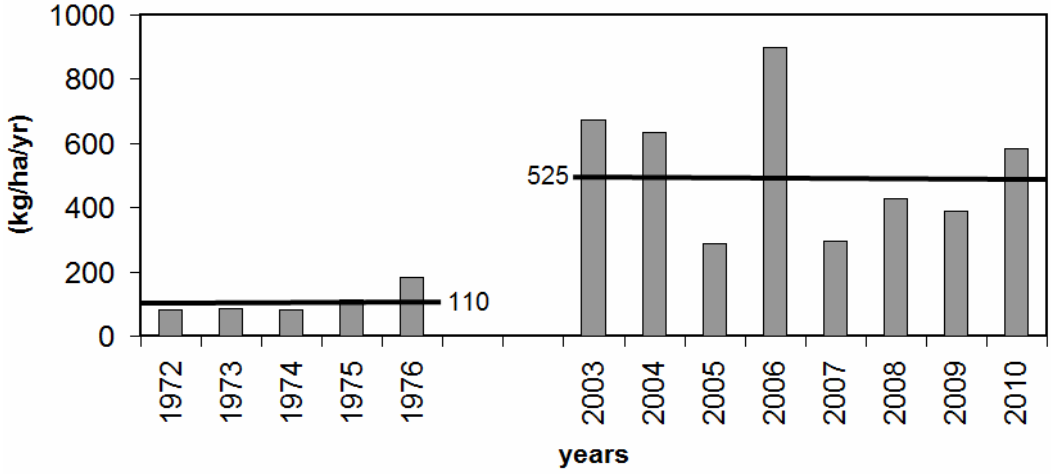



*Figure 3. Litter production of Acer campestre (with the averages) between 1972 and 2010 (kg/ha/yr)*

In Wisconsin, oaks have been more and more replaced by tree species (primarily maples) which are less sensitive to disturbance (Rogers et al. 2008). Increasing abundance and spread of the disturbance-tolerant sugar maple (*A. saccharum*) has been observed throughout the American sclerophyllous forests (Galbraith – Martin 2005). If these trends continues, disturbance-tolerating maples (*A. rubrum* and *A. saccharum*) may become the dominant species of the aforementioned forest associations (Galbraith – Martin 2005, Nowacki – Abrams 2008), because oaks cannot compete successfully against these species (McDonald et al. 2002, Zaczek et al. 2002).

Total leaf-litter production of Síkfőkút Project site (*Figure 4*) decreased as compared to the former period, differences between the two periods were less significant ($F = 4.83$, $p = 0.05$). The average leaf-litter production was 3547 kg $ha^{-1}year^{-1}$ between 2003 and 2010 which is close to the average data (4063 kg $ha^{-1}year^{-1}$) of the years 1972–1976. Thus, a small decrease can be observed in the total leaf-litter production. *Quercus cerris* and *Acer campestre* could partly compensate the deficiency of leaf-litter production which resulted from the significant mortality of *Quercus petraea*.

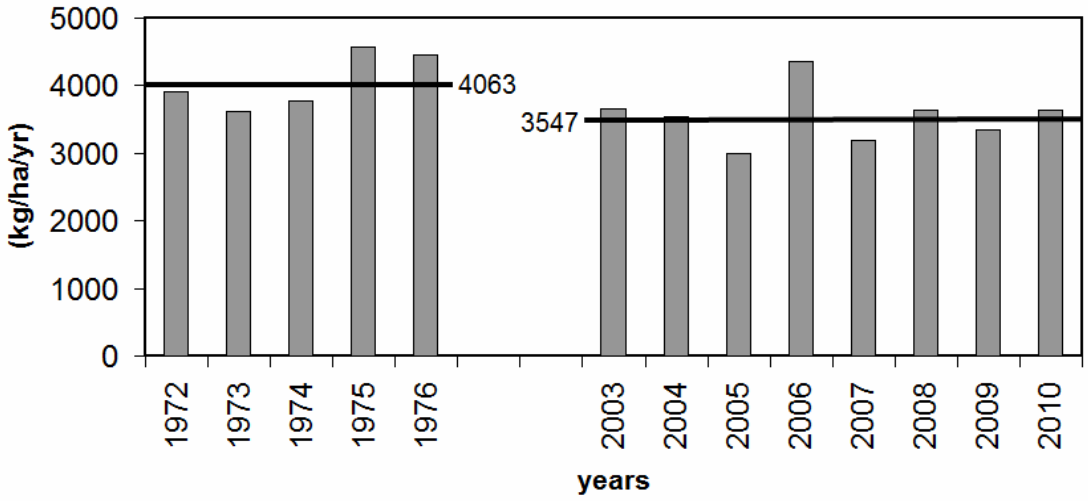



*Figure 4. Total litter production of Síkfőkút Project (with the averages) between 1972 and 2010 (kg/ha/yr)*

## 4 CONCLUSIONS

In Hungary, temperature increase was in the average 0.68 °C, while decrease of precipitation was 83 mm during the last century. As a consequence of climatic changes, the changes in structure and composition of the Síkfőkút forest are followed by the long-term changes of leaf-litter production. According to our examinations, leaf-litter production of *Quercus petraea* between 2003 and 2010 became less than a half of the amount which was measured in the earlier period (1972–1976). Leaf-litter production of *Quercus cerris* slightly increased, while leaf-litter production of *Acer campestre* multiplied. Total amount of leaf-litter production slightly decreased. In the long term, we could not detect a drastic change in the amount of leaf litter, however the qualitative composition changed. The question is whether the litter productivity will be maintained against further climatic changes, leading to further warming and aridification.

**Acknowledgments:** We wish to thank to the University of Debrecen, Department of Ecology for support the cost of field-work. Special thanks to Koncz Csabáné (Muci), Richard D. Bowden, Emese Soltész, Ágnes Nagy and Ildikó Kamenyiczki for theirs help and for the long hours of field work.

## REFERENCES

BÉRES, CS. – FENYVESI, A. – RASCHI, A. – RIDDER, H. W. (1998): Field experiment on water transport of oak trees measured by computer tomography and magnetic resonance imaging. Chemosphere 36(4–5): 925–930.

BOWDEN, R. D. – NAGEL, L. – KOTROCZÓ, ZS. – KRAKOMPERGER, ZS. – PAPP, M. – TÓTH, J. A. (2006): Long-term change in vegetation composition and biomass in a Central European oak forest at the Sikfokut International Long-Term Ecological Research (ILTER) Site, Hungary. Mid-Atlantic Ecology Conference. „Ecology in the Field" April 8–9, 2006. New Jersey, MAESA p. 2–3.

BRASIER, C. M. – ROBREDO, F. – FERRAZ, J. F. P. (1993): Evidence for *Phytophthora cinnamomi* involvement in Iberian oak decline. Plant Pathol 42: 140–145.

BRUCK, R. I. – ROBARGE, W. P. (1988): Change in forest structure in the boreal montane ecosystem of Mount Mitchell, North Carolina. Eur J For Pathol 18: 357–366.

CHAPPELKA, A. H. – FREER-SMITH, P. H. (1995): Predeposition of trees by air pollutants to low temperatures and moisture stress. Environmental Pollution 87: 105–117.

FEKETE, I. – VARGA, CS. – KOTROCZÓ, ZS. – TÓTH, J. A. – VÁRBIRÓ, G. (2011): The relation between various detritus inputs and soil enzyme activities in a Central European deciduous forest. Geoderma 167–168: 15–21.

GAERTIG, T. – SCHACK-KIRCHNER, H. – HILDEBRAND, E. E. – WILPERT, V. K. (2002): The impact of soil aeration on oak decline in south-western Germany. Forest Ecology and Management 159: 15–25.

GALBRAITH, S. L. – MARTIN, W. H. (2005): Three decades of overstory and species change in a mixed mesophytic forest in eastern Kentucky. Castanea 70: 115–128.

HAASE, R. (1999): Litterfall and nutrient return in seasonally flooded and non-flooded forest of the Pantanal, Mato Grosso. Brazil Forest Ecology and Management 117(1–3): 129–147.

HELAMA, S. – LÄÄNELAID, A. – RAISIO, J. – TUOMENVIRTA, H. (2009): Oak decline in Helsinki portrayed by tree-rings, climate and soil data. Plant and Soil 319: 163–174.

JAKUCS, P. (1985): Aims and preparation of research. *In*: JAKUCS, P. (ed.): Ecology of an Oak Forest in Hungary. Result of ′Síkfőkút Project′. Akadémiai Kiadó, Budapest.

JOLÁNKAI, M. – LÁNG, I. – CSETE, L. (2004): Hatások és alkalmazkodás. [Impacts and adaptation] Természet Világa 135: 16–18. (in Hungarian)

KLEIN, R. M. – PERKINS, T. D. (1987): Cascades of causes and effects of forest decline. Ambio 16: 86–93.

KOTROCZÓ, ZS. – KRAKOMPERGER, ZS. – KONCZ, G. – PAPP, M. – BOWDEN, R. D. – TÓTH, J. A. (2007): A Síkfőkúti cseres-tölgyes fafaj összetételének és struktúrájának hosszú-távú változása. [Long term changes in the compositionand structure of an oak forest at Síkfőkút, North Hungary] Természetvédelmi Közlemények 13: 93–100. (in Hungarian with English summary)

KOVÁCS-LÁNG, E. – HERODEK, S. – TÓTH, J. A. (2000): Long-Term Ecological Research in Hungary. In: The International Long-Term Ecological Research Network. Perspectives from Participating Networks. Compiled by the US LTER Network Office Albuquerque New Mexico, p. 38–40.

MÁTYÁS, CS. (2004): A természetes növénytakaró, az erdő klímaérzékenysége. [The natural vegetation and the forest climate sensitivity.] Természet Világa. 135: 70–73. (in Hungarian)

MÁTYÁS, CS. (2010): Forecasts needed for retreating forests. Nature 464: 1271.

MCDONALD, R. I. – PEET, R. K. – URBAN, D. L. (2002): Environmental correlates of oak decline and red maple increase in the North Carolina Piedmont. Castanea 67: 84–95.

MEENTEMEYER, V. – BOX, E. O. – THOMPSON, R. (1982): World Patterns and Amounts of Terrestrial Plant Litter Production. BioScience 32 (2): 125–128.

NOWACKI, J.G. – ABRAMS, D. M. (2008): The demise of fire and "Mesophication" of forests in the Eastern United States. Biosci. 58: 123–138.

PANDE, P.K. – MESHRAM, P. B. – BANERJEE, S. K. (2002): Litter production and nutrient return in tropical dry deciduous teak forests of Satpura plateau in central India. Tropical Ecology 43(2): 337–344.

ROGERS, D.A. – ROONEY, T.P. – OLSON, D. – WALLER, D. M. (2008): Shifts in Southern Wisconsin forest canopy and understory richness, composition, and heterogeneity. Ecology 89: 2482–2492.

SICCAMA, T. G. – BLISS, M. – VOGELMANN, H. W. (1982): Decline of red spruce in the Green Mountains of Vermont. Bull. Torrey Bot. Club 109: 162–168.

SUKUMAR, R. – DATTARAJA, H. S. – SURESH, H. S. – RAMAKRISHNAN, J. – VASUDEVA, R. – NIRMALA, N. – JOSHI, N. V. (1992): Long term monitoring of vegetation in a tropical forest in Madumali, southern India. Current Science 62: 608–616.

SZALAI, S. (2004): Igazolják-e a felmelegedést a megfigyelt adatok? [Is warming justified ba observed data?] Természet Világa. 135: 48–50. (in Hungarian)

THOMAS, F. M. – BÜTTNER, G. (1998): Nutrient relations in healthy and damaged stands of mature oaks on clayey soils: two case studies in northwestern Germany. For Ecol Manage. 108: 301–319.

THOMAS, F. M. – BLANK, R. – HARTMAN, G. (2002): Abiotic and biotic factors and their interactions as causes of oak decline in Central Europe. Forest Pathology 32: 277–307.

TÓTH, J. A. – LAJTHA, K. – KOTROCZÓ, ZS. – KRAKOMPERGER, ZS. – CALDWEL, B. – BOWDEN, R. D. – PAPP, M. (2007): A klímaváltozás hatása az elhalt szerves anyag lebontási folyamataira [The probable effect of climate change on soil organic matter decomposition]. In: Erdő és Klíma V. NyME, Sopron. pp. 307–322. (in Hungarian with English summary).

TÓTH, J. A. – PAPP, B. L. – JAKUCS, P. (1985): Litter production of the forest. In: JAKUCS, P. (ed.): Ecology of an oak forest in Hungary. Results of ′Síkfőkút Project′ 1. Akadémiai Kiadó, Budapest, 211–225.

ZACZEK, J. J. – GRONINGER, J. W. – VAN SAMBEEK, J. W. (2002): Stand dynamics in an old-growth hardwood forest in Southern Illinois, USA. Nat Areas J. 22: 211–219.

VERSITA DOI: 10.2478/v10303-012-0004-5 Acta Silv. Lign. Hung., Vol. 8 (2012) 39–55

# Soil Erosion Analysis in a Small Forested Catchment Supported by ArcGIS Model Builder

Péter CSÁFORDI[a*] – Andrea PŐDÖR[b] – Jan BUG[c] – Zoltán GRIBOVSZKI[a]

[a] Institute of Geomatics and Civil Engineering, Faculty of Forestry, University of West Hungary, Sopron
[b] Department of Geoinformation Science, Faculty of Geoinformatics, University of West Hungary, Székesfehérvár
[c] Institute of Physical Geography and Landscape Ecology, Faculty of Natural Sciences, Leibniz University of Hannover

**Abstract** – To implement the analysis of soil erosion with the USLE in a GIS environment, a new workflow has been developed with the ArcGIS Model Builder. The aim of this four-part framework is to accelerate data processing and to ensure comparability of soil erosion risk maps. The first submodel generates the stream network with connected catchments, computes slope conditions and the LS factor in USLE based on the DEM. The second submodel integrates stream lines, roads, catchment boundaries, land cover, land use, and soil maps. This combined dataset is the basis for the preparation of other USLE-factors. The third submodel estimates soil loss, and creates zonal statistics of soil erosion. The fourth submodel classifies soil loss into categories enabling the comparison of modelled and observed soil erosion. The framework was applied in a small forested catchment in Hungary. Although there is significant deviation between the erosion of different land covers, the predicted specific soil loss does not increase above the tolerance limit in any area unit. The predicted surface soil erosion in forest subcompartments mostly depends on the slope conditions.

**GIS / forest cover / erosion modelling**

**Kivonat** – **Talajeróziós elemzések egy erdősült kisvízgyűjtőn az ArcGIS Model Builder segítségével.** A tanulmány egy új munkafolyamatot mutat be, amely az Általános Talajvesztési Egyenlet (USLE) térinformatikai környezetben való alkalmazását könnyíti meg. Az ArcGIS Model Builder-ben létrehozott négyrészes keretrendszer meggyorsítja az adatfeldolgozást és biztosítja a talajeróziós térképek összehasonlíthatóságát. Az első modul – a digitális domborzatmodellből kiindulva – előállítja a lefolyáshálózatot és a kapcsolódó vízgyűjtőket, megadja a lejtőadottságokat és az USLE LS faktorát. A második modul egyesíti a lefolyáshálózatot, az utakat, a vízgyűjtőhatárt, a felszínborítást, a területhasználatot és a talajtérképet tartalmazó vektoros rétegeket. Ez az egyesített adatbázis az alapja a többi USLE-tényező előkészítésének. A harmadik modul kiszámolja a talajveszteséget, és területi statisztikákat képez a talajerózióhoz táblázatos és térképi formában. A negyedik modul vektoros talajveszteségi térképeket konvertál, ahol az egyes poligonok megegyeznek az egyes talajveszteségi osztályokkal. Így lehetővé válik a modellezett és a terepen felmért talajerózió összehasonlítása. A keretrendszert egy hazai erdősült kisvízgyűjtőn alkalmaztuk. Habár jelentős eltérést tapasztaltunk a különböző talajborítású területek eróziója között, a megengedett talajveszteségi értéket egyik területi egységben sem haladta meg a modellezett felületi talajpusztulás. A vizsgálati területen – az egyes erdőrészleteket tekintve – a felületi talajpusztulást legfőképp a domborzati adottságok befolyásolták.

**GIS / erdőborítás / eróziómodellezés**

* Corresponding author: csafordip@emk.nyme.hu; H-9400 SOPRON, Bajcsy-Zs. u. 4.

## 1 INTRODUCTION

Water-driven erosion is a natural process which is responsible for landscape degradation. Forest vegetation generally is good for soil protection, however, human disturbances may accelerate erosion on territories with high relief energy. Climate change (e.g. including more intensive storms) is also responsible for soil loss intensification. Several studies discuss how forestry activities (Surfleet – Ziemer 1996, Lisle 1998) and land use change influence the sediment transport (Sorriso – Valvo et al. 1995, García – Ruiz et al. 2008). Removing trees reduces evapotranspiration and rainfall interception leading to increased surface runoff. Furthermore without vegetation, soils become vulnerable to surface erosion. Uprooting and road cuts may decrease slope stability and cause mass movement erosion. Heavy equipment can compact soils during roadwork, log skidding, construction. Roads and landings decrease infiltration, increase and concentrate overland flow. Linear structures alter drainage paths and redirect water to more erodible areas. The construction of stream crossings and roads are a major source of erosion in forested catchments due to the low permeability of the road surface (Lewis 1998, Chang 2006).

Soil loss leads to a decrease of the water holding capacity, nutrient availability and organic matter content and to a reduction in the overall fertility of arable lands. Siltation of streams and lakes is another consequence of soil erosion. Diminishing reservoir capacity and flow cross-section of channels may cause a higher flood risk. Aquatic habitats may change if the channel morphology changes (Shen – Julien 1993, Gordon et al. 2004, Gomi et al. 2005, Chang 2006). The eroded and then suspended sediment increases turbidity, reducing the visibility distance in the water body, and the depth at which photosynthesis takes place. Suspended sediment can directly damage fish gills, diminish drinking water quality and impair irrigation systems (Lewis 1998, Gomi et al. 2005).

### 1.1 Goals and motivations for the study

The need for multi-institutional collaboration motivated the development of this framework. The soil erosion risk of forested catchments in Hungary is being researched by the Hungarian Forest Research Institute in the Mátra Mountains (Bánky 1959, Újvári 1981), and by the University of West Hungary in the Sopron Hills (Kucsara – Rácz 1988, Gribovszki 2000, Gribovszki – Kalicz 2003, Csáfordi et al. 2010, Csáfordi 2010). Several studies from Leibniz University of Hannover have analysed the effects of soil conservation activities on agricultural plots in Lower Saxony, Germany (Mosimann et al. 2004, Sanders 2007, Bug 2011). A comparative study concerning potential soil erosion estimation with the USLE is planned in cooperation with the University of Hannover. The scopes of the study are

- to develop a workflow in the ArcGIS Model Builder which predicts soil erosion with the Universal Soil Loss Equation (USLE) in a uniform way, creates a similar type of zonal statistics based on potential soil loss, and compares the results of different study areas. This framework can be a new tool which simplifies and accelerates the soil erosion prediction for different land use practices, land cover, and rainfall scenarios related to expected climate change;
- to model the potential surface erosion in the Farkas Ditch (Sopron Hills);
- to reveal USLE-factors, which significantly influence surface erosion in the study catchment, using correlation analysis.

### 1.2 Spatially distributed physical soil erosion models and the USLE

Many models have been developed to predict areas that are susceptible to water erosion, to predict soil loss, and to evaluate soil erosion-control practices. Physically based models like WEPP (Nearing et al. 1989), EROSION-3D (von Werner 1995), or LISEM (De Roo et al. 1996) take the spatial variability of land use and hydrological processes into account and estimate soil loss based on physical laws on a watershed scale. However they often can not be used because of extensive data requirements. Many input parameters require further calibration, complex laboratory analysis or expensive field data collection (Ma 2001, Beskow et al. 2009). Lack of data may lead to unrealistic predictions (Fistikoglu – Harmancioglu 2002).

In contrast to physically based models, Martin et al. (2003) note that empirical models such as the USLE require less site specific data. Therefore the USLE is more widely applied for predicting of soil losses and for planning soil conservation measurements, especially in developing countries (Jain – Kothyari 2000, Lu et al. 2004, Onyando et al. 2005, Erdogan et al. 2007, Pandey et al. 2007). The USLE is an empirical equation originally developed by Wischmeier – Smith (1978) in the USA, where the average specific soil loss pro unit area can be computed by multiplying the following six factors:

$$A = R \cdot K \cdot L \cdot S \cdot C \cdot P \quad (1)$$

In *Eq. (1)* $A$ is the mean annual soil loss ($t \cdot ha^{-1} \cdot yr^{-1}$); $R$ is the rainfall-runoff erosivity ($kJ \cdot m^{-2} \cdot mm \cdot h^{-1}$), which represents the erosion potential of locally expected rainfalls on cultivated soil without vegetation cover. $K$ is the soil erodibility ($t \cdot ha^{-1} \cdot m^2 \cdot kJ^{-1} \cdot h \cdot mm^{-1}$). It shows the rate of soil loss per unit of rainfall for a specific soil for a clean-tilled fallow. $L$ is the length of the slope (dimensionless), the rate of soil loss compared to the soil loss from a slope 22.13 m long. $S$ represents the slope steepness (dimensionless), the rate of soil loss compared to the soil loss of a slope with a 9% inclination. $C$ is the cover-management (dimensionless), which shows the influence of plants in contrast to bare fallow. $P$ is the erosion-control practice (dimensionless). Control practices are usually contours, strip cropping or terraces (Centeri 2001, Amore et al. 2004). The calculated soil loss can be compared to the tolerable soil loss. The tolerable soil loss is the maximum level of soil erosion that still allows a high level of crop productivity over the years (Stone – Hilborn 2000, Severin et al. 2003).

Many authors have discussed the applicability of the USLE in different study areas. Originally, the USLE allows the long term prediction of soil loss only for standardised agricultural plots (Wischmeier – Smith 1978, Schwertmann et al. 1987). The adaptation of the equation to a wider scale and to different land uses, such as forests, is not recommended by Wischmeyer – Smith (1978). However, several other authors have proven that the USLE is capable of estimating soil loss on a wider scale (Jain – Kothyari 2000, Onyando et al. 2005, Khosrowpanah et al. 2007, Beskow et al. 2009). Rácz (1985) has suggested factor values for the USLE adaptation in forest lands of Hungary. Bartsch et al. (2002) have applied the empirical equation to determine the most erosion sensitive areas in a rangeland with complex topography and varied land uses such as grazing and military activities. A major problem is the value of the predicted soil loss. It can exceed the actual values by one order of magnitude in forested areas. Because the soil distribution is mostly irregular and surface runoff is often prevented by organic debris (such as logs, twigs, and sometimes leaves), the USLE overestimates the soil loss (Risse et al. 1993). The USLE was developed for the prediction of sheet and rill erosion. However the results show no separate values for rill and inter-rill erosion, but overall soil loss only. The USLE is also not feasible for estimating the amount of deposition, and for calculation of sediment yield from gully, streambed and streambank erosion (Wischmeier – Smith 1978, Fistikoglu – Harmancioglu 2002, Andersson 2010). The equation was primarily designed for calculating long-term average annual rates of erosion

(Stone – Hilborn 2000, Beskow et al. 2009). It is therefore necessary to develop techniques to estimate soil loss for individual storm events (Jain – Kothyari 2000). Andersson (2010) cites that interactions between USLE-factors are not taken into account.

### 1.3 Soil erosion models and Geographical Information Systems

Soil erosion risk differs spatially because of heterogeneous topography, geology, geomorphology, soil types, land cover, and land use. Geographical Information Systems (GIS) are able to handle these spatially variable data easily and efficiently. The estimation of soil erosion with GIS techniques reduces costs and improves accuracy (Ma 2001, Erdogan et al. 2007, Khosrowpanah et al. 2007). State-of-the-art GIS provides the necessary mapping and interpolation methods to create a database, which includes all input datasets for erosion modelling. The resolution should reflect the spatial variation of the hydrological and erosion processes (Fistikoglu – Harmancioglu 2002, Beskow et al. 2009). Decreasing cell size and increasing scale requires a large amount of data for accurate prediction. GIS is therefore most appropriate for the management of a huge amount of data. It reduces time and costs for accessing and handling a database (De Roo – Jetten 1999). De Roo et al. (1996), Fistikoglu – Harmancioglu (2002), Khosrowpanah et al. (2007), and Pandey et al. (2007) describe even more advantages of GIS, such as the production of complex input maps and the combination of soil, land use and land cover information. With GIS techniques, the calculation of soil loss rates for alternative land management scenarios becomes easier.

The required data for the prediction of soil loss (rainfall erosivity, soil data, digital elevation model and land use) has to be converted into a GIS-format in order to implement the USLE in GIS. Different authors have used GIS-based techniques to model USLE-factors for predicting soil loss for larger watersheds on a grid cell basis (Erdogan et al. 2007, Andersson 2010). According to Martin et al. (2003), a combined USLE/GIS approach is able to identify discrete locations with precise spatial boundaries with a high erosion potential. Beskow et al. (2009) validate that the combined USLE/GIS technique shows an acceptable accuracy and allows mapping of the most susceptible areas. The studies by Onyando et al. (2005) and Erdogan et al. (2007) contradict this. The upscaling of the USLE-applications from plots to large watersheds is limited depending on the reliability and availability of direct field measurements. As Fistikoglu – Harmancioglu (2002) cite, the results of erosion risk assessment are more plausible for small grid sizes and smaller areas. Therefore larger watersheds must be analysed as sub-basins. A comprehensive USLE/GIS application was accomplished in Balaton Project in Hungary, where Kertész et al. (1992, 1997) have divided the Örvényesi watershed into “erotopes” which are “inclined parts of the relief with an unconcentrated runoff in more or less the same direction” (Kertész et al. 1997 p. 22). This technique makes it possible to analyse the impact of unconcentrated runoff and to model soil erosion in a larger catchment at quasi-plot scale or in slope segments.

The combined USLE/GIS approach is also limited by each input factor. Auerswald (1987) states that the calculated soil loss is highly sensitive to the slope. Modern GIS-based procedures support the calculation of other USLE-factors as well. Many studies applied remote sensing data to develop values for the *C* and *P* factors, to classify land cover categories and land use units (Ma et al. 2003, Beskow et al. 2009). These studies confirm that the original spatial limitations of the USLE can be avoided by using remote sensing data and GIS. Márkus – Wojtaszek (1993a,b) have conducted the USLE calculation in an ArcInfo environment and compared the density differences of aerial photographs and satellite images with the erosion sensitive areas. The results prove that remote sensing is a suitable method to check the modelled soil erosion categories and to follow the actual stage of the erosion processes. The integration of GIS-based techniques into the USLE is useful to describe areas that are vulnerable to soil erosion, enabling immediate conservation planning (Lee 2004, Beskow et al. 2009).

## 2 MATERIAL AND METHODS

The GIS framework which is introduced in this paper helps predict surface soil erosion. It has been developed for a forested catchment (Farkas Ditch, 0.6 km$^2$) in the Sopron Hills. Soil erosion occurs here on cutting areas and unpaved forest roads. Landslides, streambank, streambed and gully erosion are also observed in the study area.

The framework combines the pre-processing of digital data and different geoprocessing tools in the ArcGIS/ArcMap 9.3 environment to generate the required factors to predict soil erosion. In the Farkas Ditch, the input data consists of a 5x5 m raster resolution DEM (FÖMI, DDM-5), a 1:10000 scale digital topographic map (FÖMI, DTA-10), aerial photographs (FÖMI, "Aerial Measurement of Hungary 2008"; resolution 0.5 m), forest management plans (ÁESZ 1994, 2004), a soil map based on the analysis of soil samples from the Farkas Ditch, and a survey of the eroded areas. The DEM is the input for modelling the catchment boundary, the stream network, and the *LS* factor. A specific threshold is needed to model the stream network. A grid cell is considered to be a channel if the catchment above the point is greater than the specific threshold (Jain – Kothyari 2000). Land cover types, land use units, and roads of the catchment were digitalised on the basis of topographic maps, forest management plans, and aerial photographs.

### 2.1 Factors of the USLE

Land units in the vector layers such as land cover and land use maps provide the spatial distribution of the six USLE factors. To integrate the USLE in an ArcGIS/ArcMap environment, each factor must be available as a thematic raster layer. Therefore vector datasets must be converted into a grid format with the same raster resolution as the DEM. The USLE-calculation is a raster-based function, where the model multiplies the unique value of each spatially corresponding grid cell in the six thematic raster layers based on the *Eq. (1)*. The model output is the average annual soil loss (Andersson 2010). The *R* factor in this study is determined on the basis of rainfall data recorded in the 2008–2009 hydrological year in the Hidegvíz Valley rain gauge station (Sopron Hills). The *K* factor is estimated using a dataset of grain size analysis, water content measurement, and organic substance analysis of 25 soil samples collected from the upper 50 cm of the soil layer in the Farkas Ditch. The *C* factor is derived from the literature according to Ma (2001) and US EPA (2009). Recommendations by Rácz (1985) based on the tree harvesting and planting techniques were applied to define the *P* factor. The *C* and *P* factors are locally modified on the basis of field experience, forestry management plans, and visual interpretation of aerial photographs. Our previous papers (Csáfordi et al. 2010, Csáfordi 2010) describe the determination of the USLE factors in more detail.

The *LS* factor is based on the DEM and the unit stream power theory of Moore – Burch (1986). The following *Eq. (2)* was applied:

$$LS = \left( FlowAccumulation \cdot \frac{CellSize}{22.13} \right)^{m} \cdot \left( \frac{\sin \beta}{0.0896} \right)^{n} \qquad (2)$$

In *Eq. (2) Flow Accumulation* is a raster layer representing the upslope cell number contributing to the surface runoff of a certain raster cell; *Cell Size* refers to the resolution of the DEM; *m* and *n* are empirical exponents. Due to the lack of detailed digital elevation data we applied the values $m = 0.4$ and $n = 1.3$, in correspondence to other international studies such as Lee (2004) and Demirci – Karaburun (2011). Values of *m* and *n* have been suggested by Moore – Burch (1986) for standard reference conditions of USLE, where the slope-length is 22.13 m and slope is 9%.

The tolerable soil loss values according to Rácz (1985) enabled the determination of the most sensitive areas to soil erosion. Limit values at different soil depths are: 1 t/ha/yr at 20 cm, 2.2 t/ha/yr at 40 cm, 4.1 t/ha/yr at 60 cm, 6.4 t/ha/yr at 80 cm, 9 t/ha/yr at 100 cm, 11.8 t/ha/yr at 120 cm, 15.0 t/ha/yr at 140 cm. According to the forestry management plan the average soil depth in the Farkas Ditch is between 60 and 100 cm. However the field assessment shows that the soil depth can be lower or higher if data with a higher spatial resolution is available. Therefore each tolerance category is applied to enable the creation of erosion risk maps for other catchments and local heterogeneities.

### 2.2 The Model Builder function of the ArcGIS/ArcMap 9.3

The ArcGIS/ArcMap Model Builder combines several GIS operations and runs these modules with different datasets (Pfaff – Glennon 2004). A model consists of three fundamental elements: input parameters, geoprocessing tools, and output data. Model parameters are specific model inputs which need to be defined by the user. For example, the user has to define the specific location of input data, or has the opportunity to specify thresholds. Geoprocessing tools produce output data in a user-defined sequence using the input datasets.

## 3 RESULTS

This chapter describes the framework developed for surface erosion analysis with the USLE in the ArcGIS environment, the predicted surface erosion in the Farkas Ditch (Sopron Hills, Hungary), and the results of correlation analysis which reveals the most important factors influencing surface erosion in the study catchment.

### 3.1 Development of the workflow “Erosion analysis” in the ArcGIS Model Builder

The framework “Erosion analysis” consists of the four submodels: “Relief analysis”, “Soil and land cover”, “Soil loss and statistics”, and “Regionalisation”.

#### *3.1.1 The submodel “Relief analysis”*

The submodel “Relief analysis” generates a flow accumulation grid, the channel network with connected catchments, and computes slope features and the *LS* factor. *Figure 1* presents the conceptual flow chart of the first submodel. The blue ellipses mark the input model parameters, while green ellipses are the input layers.

In the first section of the model run, the “Flow Direction” raster is produced after correcting gridding artefacts. The “Flow Direction” contains the preferred direction of flow of each cell, and provides the basis of the “Flow Accumulation” raster, namely the accumulated flow to each cell. The user must provide the *threshold area* for channel initiation in order to create an adequate “Channel Network”. The threshold area in our study is based on a visual trial, a comparison between the results of testing different values and the real channel network observed in the field. The modelled channel network coincides with the real geographical conditions of the Farkas Ditch when the threshold area is 12,500 m². The model generates the “Catchment boundaries”, the selection basis of the study catchment according to the “Channel network”. Catchment boundaries and channel networks are available as raster layers at first, therefore the model converts them from raster layers into features in order to support the work with separated polylines and polygons during the following steps of erosion analysis. The modelled stream networks and catchment boundaries are only usable if the DEM supplies reliable outputs, therefore they

have to be checked by the user. In our case study these layers were generated inaccurately and had to be digitized manually. The last steps of the first phase of the "Relief analysis" submodel are the calculations of slope conditions in degrees and percent rise.

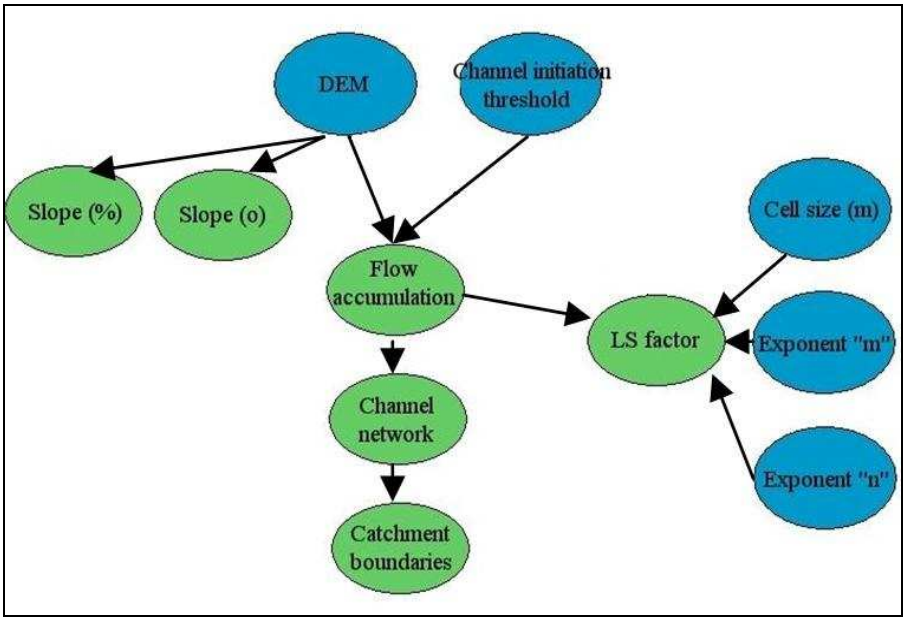



*Figure 1. Conceptual flow chart of the first submodel "Relief analysis"*

*Eq. (2)* describes the second part of the "Relief analysis", where the *LS* factor is computed using the parameters "Slope ($^{o}$)" and "Flow accumulation", taken from the first part of the submodel. *Eq. (2)* needs three other inputs, such as "Cell size", "Exponent *m*" and "Exponent *n*". The cell size is 5 m according to the raster resolution of the DEM, and the exponent values are given in Chapter 2.1. For the calculation of *LS* factor, the terrain was simplified, and the slope-length modifying effect of artificial linear elements such as roads and ditches was neglected.

### *3.1.2 The submodel "Soil and land cover"*

The feature layers *catchment boundary, channel network, roads, land cover types, land use,* and *soil map* are the result of pre-processing. The parameter "Land cover" includes the digitized land cover/vegetation categories. The layer "Land use" consists of polygons of different land use units, such as forest subcompartments and plots. "Soil types" shows the soil map which is based on physical soil properties. Linear spatial elements are represented as "Roads" and "Channel network". "Catchment boundary" refers to the borders of the study area. The submodel "Soil and land cover" (*Figure 2*) integrates these input layers and generates a feature layer ("Full soil and land cover dataset") containing all spatial information for uploading the USLE factors which are manually calculated.

The unpaved roads are mostly damaged by gully erosion caused by concentrated runoff in the experimental catchment. Since normally the impact of unconcentrated runoff is modelled by the USLE (Kertész et al. 1997), erosion calculation has not been conducted with the USLE on the surface of dirt roads and channels. Therefore all linear elements such as roads and channel networks must be removed from the study area. In the ArcGIS geoinformatical software, the vector layers "line type" do not have width. If linear elements need to be handled as areas in order to be erasable from the combined layer ("Union of land cover, land use, soil types, roads and channels,") width has to be added to the lines. Therefore the lines are buffered with a half cell size buffer distance (2.5 m in this study),

consequently the lines can be rasterised and erased for subsequent USLE-calculations. After this step, roads and channels were replaced as hollows ("Land cover, land use and soil dataset without the area of linear elements.") Surface erosion modelling is not interpreted in these areas.

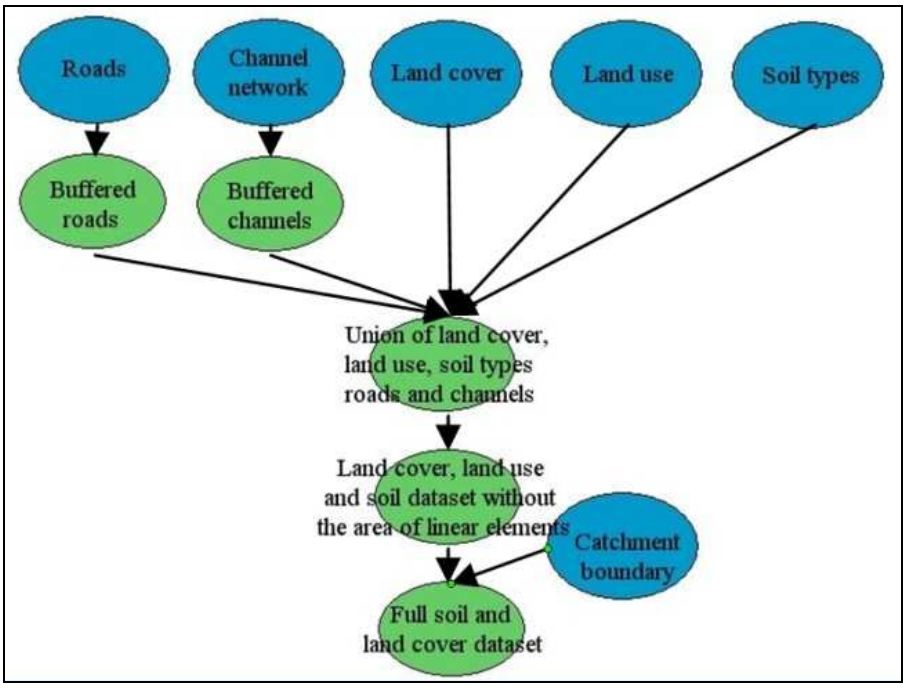



*Figure 2. Conceptual flow chart of the second submodel "Soil and land cover"*

The final section of the second submodel removes unnecessary attributes of the "Full soil and land cover dataset", as defined by the parameter "Drop Fields". This step provides the layer "Full soil and land cover", which shows the complete spatial database combination and the final structure of the attribute table for the soil erosion analysis. The attributes are: Object ID, Soil type, K factor, Land use unit, Land use practice, P factor, Code number of land cover category, Land cover category, C factor, Polygon area and R factor.

The *R* factor values are calculated with MS Excel. The *C* and *P* factors have to be filled in manually in the attribute table based on visual interpretation of aerial photographs and field experience. Because of this drawback the workflow is recommended principally for catchments smaller than 1 $km^2$. The benefit of the submodel is that several small polygons are produced with multiplied intersections, and different factor values can be given for each small polygon. This leads to a higher spatial resolution and to a more precise prediction of soil loss.

### *3.1.3 The submodel "Soil loss and statistics"*

The submodel "Soil loss and statistics" (*Figure 3*) computes the potential soil loss, evaluates and summarizes soil loss and elevation data in each land use unit and land cover category. The parameter "Full soil and coverage" contains the attribute fields of *R*, *K*, *C* and *P* factors. These attributes are converted into separate thematic raster layers, using the cell size determined by the "DEM". The model multiplies the rasterised USLE factors on a cell-by-cell basis using the *Eq. (1)*, resulting in the potential specific annual soil loss by surface erosion for each cell.

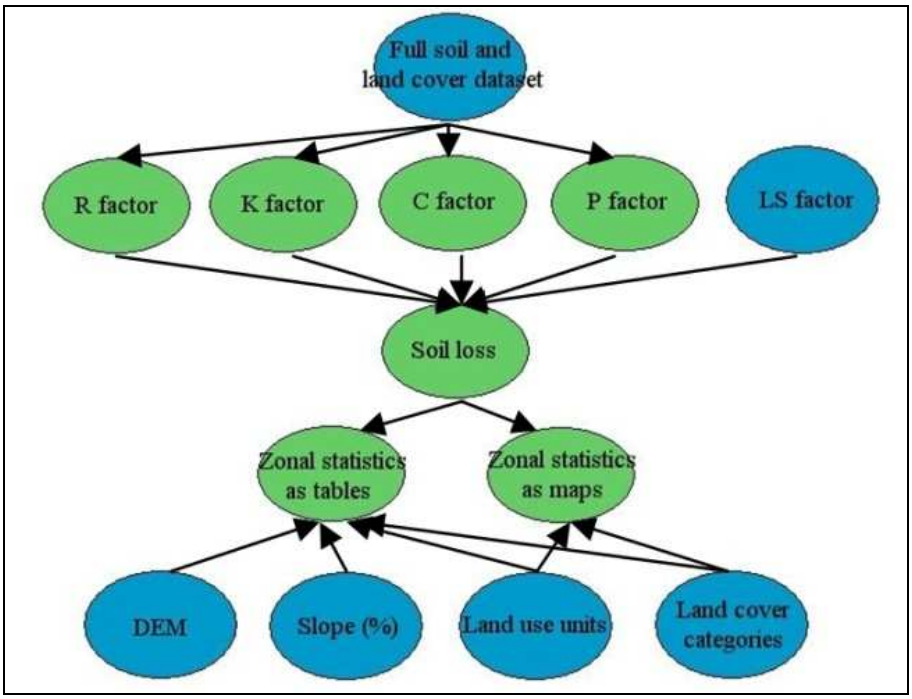



*Figure 3. Conceptual flow chart of the third submodel "Soil loss and statistics"*

The "DEM", "Slope (%)", "Land use units" and "Land cover categories" generated by the second submodel are the input layers for statistical analysis. The tool "Zonal Statistics as Tables" summarizes the values of a raster, such as elevation data, percentage slope conditions and soil loss, within the zones of another dataset (land use units and land cover/vegetation types). Tables contain the area of the polygons in the zone dataset, maximum, minimum, range, mean, standard deviation and sum. The tool "Zonal Statistics as Maps" reports the average and total soil loss for each land use category and land cover type as a raster map.

### *3.1.4 The fourth submodel "Regionalisation"*

To enable an area-based comparison between predicted and measured soil erosion, the calculated soil loss is converted from a raster into vector format. This allows:

- the comparison of the location of potential and real eroded surfaces,
- the comparison of the size of modelled and observed eroded surfaces.

The submodel "Regionalisation" converts a classified raster layer of soil loss into polygons, keeping the soil loss categories of Rácz (1985). The input soil loss raster has to be multiplied by a given constant value 10 in the first section of model run, in order to avoid conversion errors at raster values smaller than 1. Conditional if/else evaluation on each cell of input raster is applied to select distinct soil loss classes, which are the basis of generating separated polygons. The last steps integrate polygons of each soil loss category and calculate their area in $m^2$. *Figure 4* shows the raster map of modelled soil erosion on the left and the soil loss classes as polygons after conversion on the right.

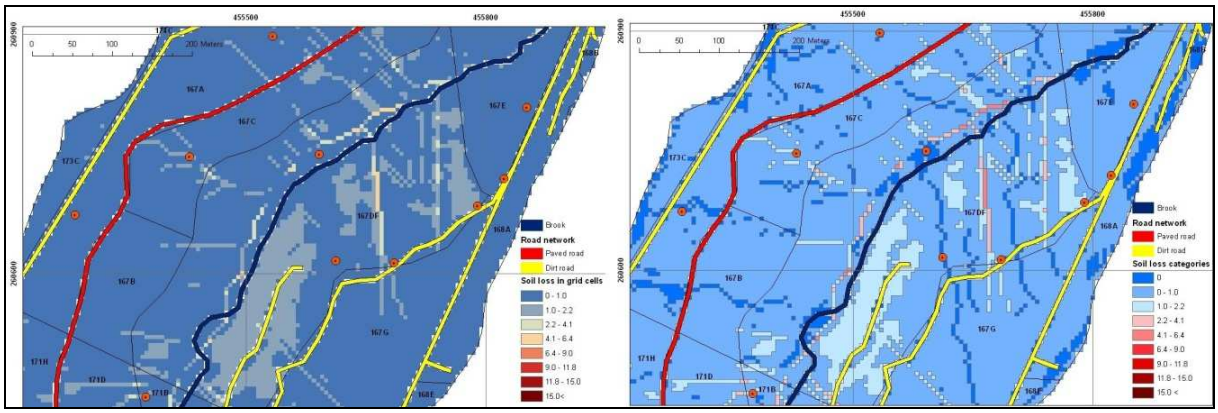



*Figure 4. The raster map of modelled soil erosion and polygons after conversion*

### 3.2 Soil erosion prediction in the Farkas Ditch

A surface soil erosion scenario was modelled with the ArcGIS Model Builder workflow for the hydrologic year 2008–2009 in the Farkas Ditch using the USLE factors shown in *Table 1*.

*Table 1. USLE factors and predicted soil loss*

| | Factors of the USLE | | | | | |
|---|---|---|---|---|---|---|
| | $R$ (kJ·m$^{-2}$·mm·h$^{-1}$) | $K$ (t·ha$^{-1}$·m$^{2}$·kJ$^{-1}$ ·h·mm$^{-1}$) | $LS$ | $C$ | $P$ | $A$ (t·ha$^{-1}$·yr$^{-1}$) |
| Value / Interval | 108.4 (constant) | 0.32–0.42 | 0–95.6 | 0.003–0.01 | 0.2–0.4 | 0–6.1 |
| Mean | – | 0.36 | 6.9 | 0.006 | 0.24 | 0.5 |
| SD | – | 0.09 | | 0.002 | 0.08 | 0.5 |

*Figure 4* presents a part of the soil erosion risk map with the most endangered zones. *Figure 5* shows the percent area of each soil loss category according Rácz (1985) out of the total catchment area, proving that the predicted surface erosion does not exceed the 6.4 t·ha$^{-1}$·yr$^{-1}$ (the limit value at 80 cm soil depth) in any grid cell. Moreover surface erosion remains below 2.2 t·ha$^{-1}$·yr$^{-1}$ (82.89%) in the Farkas Ditch. The total predicted soil loss is 26.4 tons from the 0.56 ha area of the Farkas Ditch without roads and channels.

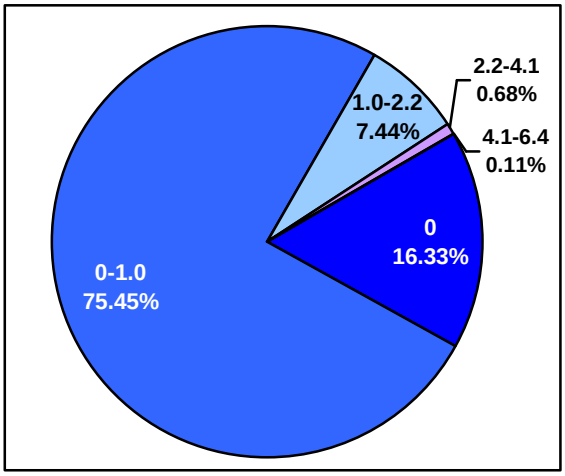



*Figure 5. Percent area of the soil loss categories*

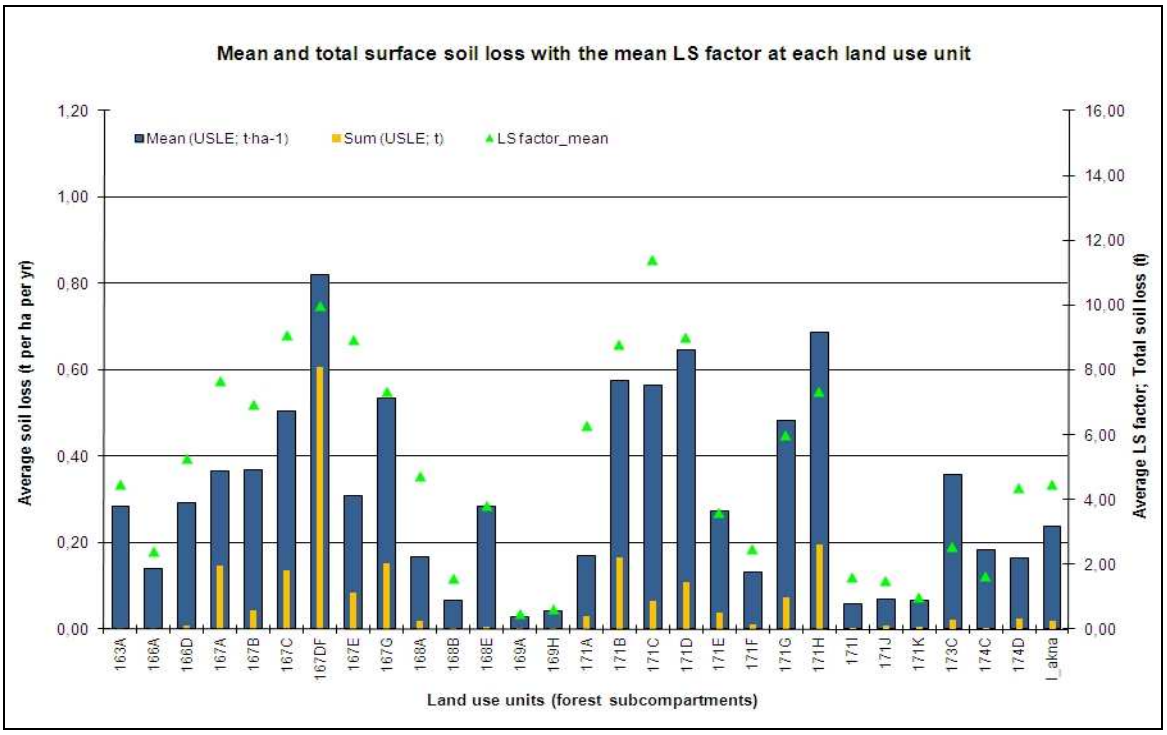



*Figure 6. Mean and total surface soil loss with the mean LS factor for each land use unit*

Mean and total surface erosion have been calculated for each land use unit and land cover type as given in *Figures 6* and *7*. The predicted soil loss does not rise above the tolerance limit in any area unit, but spatial variability of the erosion risk can be observed. The triangles represent the mean LS factor in each area unit, revealing that surface erosion risk has a significant correlation with the slope-length conditions in the forest subcompartments. Nevertheless, the *LS* factor does not account for the fluctuations of mean soil loss in the different land cover types.

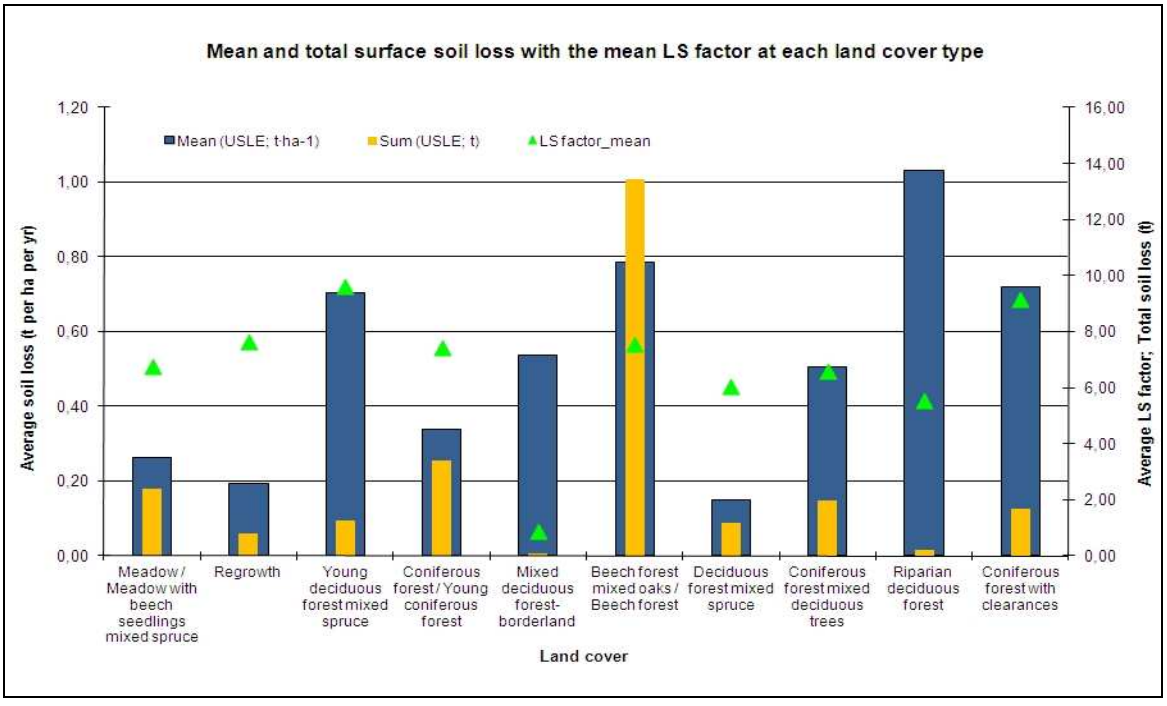



*Figure 7. Mean and total surface soil loss with the mean LS factor at each land cover type*

The preliminary assumption was that the lowest erosion risk is in forested zones, however the model results and field experience partly contradicted this hypothesis: e.g. “riparian deciduous forest” and “beech forest mixed oaks / beech forest” show the highest mean soil loss from steep slopes, the landslides next to the stream network, the sparse canopy closure, undergrowth and litter layer. However the highest mean soil loss value is also six times lower than the tolerance limit, emphasizing the soil protection role of forest vegetation.

### 3.3 USLE-factors influencing surface soil erosion in the Farkas Ditch

To assess the most determining factor for potential surface erosion in the Farkas Ditch, a site-specific correlation analysis was conducted. The results are shown in *Table 2*. Marked correlations (**bold values**) are significant at $p < 0.05$. The *C* factor is obviously the most important factor for the mean specific soil loss at the land cover categories, and the size of each unit has the most significant correlation with the total surface soil loss. More factors show significant correlations with the soil erosion on the land use units, and the *LS* factor has the strongest influence aside from the unit area.

Linear elements such as roads and channels were erased from the combined land use and land cover map, so the surface erosion analysis with the USLE was not interpreted in the areas of roads. To evaluate the soil detachment from dirt roads and skid trails, the physically distributed model EROSION-3D was applied in our previous paper (Csáfordi 2010). Calculations with EROSION-3D indicated that the mean soil loss from unpaved forest roads is six times higher than the average soil loss per land use unit. Consequently linear erosion has a higher importance as an erosion source in the Farkas Ditch than surface erosion.

*Table 2. Correlation between soil loss and different factors influencing surface erosion*

| | Area | *LS*-mean | *LS*-max | *K*-mean | *K*-max | *C*-mean | *C*-max | *P*-mean | *P*-max |
|---|---|---|---|---|---|---|---|---|---|
| Land cover categories | | | | | | | | | |
| Soil loss-mean | 0.00 | 0.06 | –0.18 | 0.04 | –0.22 | **0.69** | 0.38 | 0.41 | –0.49 |
| Soil loss-sum | **0.95** | 0.24 | 0.22 | 0.06 | 0.23 | 0.19 | 0.23 | 0.21 | 0.30 |
| Land use units | | | | | | | | | |
| Soil loss-mean | **0.72** | **0.86** | **0.77** | **0.50** | **0.64** | 0.31 | 0.29 | 0.22 | 0.23 |
| Soil loss-sum | **0.94** | **0.62** | **0.72** | 0.26 | **0.49** | 0.19 | 0.26 | 0.17 | 0.16 |

## 4 DISCUSSION

A number of international studies describe USLE/GIS implementation, and our ArcGIS workflow provides evidence that erosion modelling with the USLE can be adapted to a GIS-environment. Producing thematic raster layers of USLE-factors in GIS and calculation of soil erosion using them is discussed among others by Kertész et al. (1992, 1997), Márkus – Wojtaszek (1993a,b), Jain – Kothyari (2000), Lu et al. (2004), Onyando et al. (2005), Erdogan et al. (2007), Pandey et al. (2007) and Beskow et al. (2008). Khosrowpanah et al. (2007), Andersson (2010) and Demirci – Karaburun (2011) have also performed their analyses within an ArcGIS/ArcMap framework. The workflow of Khosrowpanah et al. (2007) can achieve a more accurate prediction, because a C++ executable program (Van Remortel et

al. 2004) computes the *LS* factor for each grid cell of the DEM input. Our research is limited to erosion prediction, but Jain et al. (2010) have calculated sediment yield and deposition besides soil loss, using the spatially distributed sediment transport capacity.

Although we do not have direct field measurements of surface soil erosion, a total sediment load has been applied to verify the reliability of the predicted soil loss. According to our previous study (Csáfordi et al. 2011) the total sediment yield is 95.1 tons in the hydrologic year 2008–2009 in the stream of Farkas Ditch. The total surface soil loss reduced the sediment delivery ratio by 50% (Csáfordi et al. 2010). There is a 13.9% portion (13.2 tons) from the annual sediment load, and a depleting sediment deposit behind a log jam has a 16.6% (15.8 tons) contribution to the sediment load. Surface erosion does not represent the major part of sediment resources in the Farkas Ditch in the reference period, therefore other erosion phenomena, such as mass movement and channel erosion, have to be calculated. Lee et al. (2004) have compared the surface erosion potential map with landslide location data and found that many landslides occurred where the *LS* factor is 0 and the soil loss value is 0. This fact draws our attention to the possible errors of the *LS* factor calculating process, to the requirement of DEM with a higher raster resolution, and that it is not sufficient to evaluate surface and linear erosion in the Farkas Ditch where landslides are frequent.

Erdogan et al. (2007) have demonstrated the same results as our study in the Kazan watershed, Turkey, that soil erosion potential of the poorly managed pastures was lower as in the land of the dense forest due to relatively higher *C* values. Furthermore, the topographical properties of the watershed had a greater influence on the magnitude of soil loss than land use/land cover types. The significance of slope conditions has also been confirmed by Demirci – Karaburun (2011), where 73% of the mostly agricultural Buyukcekmece Lake watershed had low and slight erosion risks with values under 3 $t \cdot ha^{-1} \cdot yr^{-1}$. The majority of land with low and slight soil erosion risks has slopes <5%. Nevertheless, predicted surface erosion remains below 2.2 $t \cdot ha^{-1} \cdot yr^{-1}$ on 82.9% of the Farkas Ditch, in the Sopron Hills, while this rate was only about 60% in the Kazan watershed. There is a significant difference in the judgement of erosion tolerance limits according to the soil depth, because Erdogan et al. (2007) mark the >1 $t \cdot ha^{-1} \cdot yr^{-1}$ soil loss as an irreversible change, whereas soil loss below 4.1 $t \cdot ha^{-1} \cdot yr^{-1}$ means tolerable risk in our catchment (Rácz 1985).

## 5 CONCLUSION AND SUMMARY

This study describes a new implementation of the Universal Soil Loss Equation (USLE) in a GIS-environment using the ArcGIS Model Builder. The four-part combination of geoprocessing tools unifies the surface soil erosion prediction in small catchments and accelerates the working process. Nevertheless further studies are required to ensure the comparability of result structures (attribute tables, statistical values and maps) and the model efficiency, when the workflow is applied to evaluate potential soil erosion in different catchments with different land use and rainfall scenarios. Plans to apply the model in other forested catchments in Hungary and agricultural catchments in Lower Saxony have been made to further improve and to expand the model. With this application the spatial transferability of the workflow will also be analysed.

This paper describes a new aspect of GIS-supported erosion analysis in the field of forestry. Surface soil loss is computed with a cell-by-cell multiplication of six factors derived from rainfall records, soil maps, DEM, land cover and land use data. The first submodel “Relief analysis” delineates the catchments, produces the stream network, and computes the slope conditions and the LS factor. The submodel “Soil and land cover” combines land cover, land use and soil database into one layer generating an attribute table

with *R*, *K*, *C* and *P* factors. The third submodel "Soil loss and statistics" calculates soil loss using rasterised USLE-factors from the attribute table of the input layer. The submodel aggregates and calculates a statistical analysis of elevation, slope and soil loss values as a table or as a map for land use units and land cover/vegetation categories. The submodel "Regionalisation" converts the raster layer of potential soil loss to polygon keeping the erosion risk categories set in the raster theme. The submodel computes the area of erosion classes enabling the comparison of predicted erosion to field data.

A surface erosion scenario for the hydrologic year 2008–2009 was modelled in a small forested catchment, Farkas Ditch in the Sopron Hills, using the developed ArcGIS model. Although there is significant deviation between the erosion of different land cover, the predicted soil loss does not rise above the tolerance limit in any area unit. Regenerated areas with dense grass cover also have a significant soil protection function, because neither the mean nor the maximum surface soil loss exceeds the limit value. The predicted soil erosion mostly depends on the slope conditions in the forest subcompartments.

To confirm the results of USLE-evaluations, direct field measurements of surface soil erosion are required. Some improvements are projected for the future in order to achieve more reliable prediction results for smaller catchments and to extend the framework for larger watersheds. More precise automatic delineation of stream networks and catchment boundaries can be ensured by application of detailed DEM involving artificial linear elements, such as roads and ditches, which modify the slope-length and rainfall runoff. To obtain an accurate LS factor, real slope-length has to be considered, and different calculation techniques should be used. Manual uploading of cover-management and erosion-control practice factors can be automatized. But this operation may lead to a decrease of the spatial resolution of factor values, because the user has no control over the setting of the *C* and *P* factors.

**Acknowledgements:** This research was supported by the TÁMOP 4.2.1/B-09/KONV-2010–0006 project, the scholarships "Aktion für Österreich-Ungarn" and "Deutsche Bundesstiftung für Umwelt". This publication was conducted in the frame of the TÁMOP 4.2.2. B – 10/1 – 2010 – 0018 project (title: Talentum – Development of the talent maintenance conditions in the University of West Hungary) supported by the European Union with the co-financing of the European Social Fund. The authors would like to acknowledge the support from the members of Leibniz University of Hannover, Institute of Physical Geography and Landscape Ecology, Institute of Geomatics and Civil Engineering, Faculty of Forestry, University of West Hungary, and Department of Geoinformation Science, Faculty of Geoinformatics, University of West Hungary.

## REFERENCES

AMORE, E. – MODICA, C. – NEARING, M.A. – SANTORO, V.C. (2004): Scale effect in USLE and WEPP application for soil erosion computation from three Sicilian basins. J. Hydrol. 293: 100–114.

ANDERSSON, L. (2010): Soil loss estimation based on the USLE/GIS approach through small catchments. A minor field study in Tunisia. Technical report TVVR 10/5019. Division of Water Resources Engineering, Department of Building and Environmental Technology, Lund University. 5–14.

AUERSWALD, K. (1987): Sensitivität erosionsbestimmender Faktoren. Wasser und Boden 39 (1): 34–38.

BÁNKY, GY. (1959): A kisnánai eróziómérő állomás három évi munkásságának eredményei. [Three-year results of the soil erosion gauging station in Kisnána.] Erdészeti Kutatások 6 (3): 139–160. (in Hungarian)

BARTSCH, K.P. – VAN MIEGROET, H. – BOETTINGER, J. – DOBROWOLSKI, J.P. (2002): Using empirical erosion models and GIS to determine erosion risk at Camp Williams, Utah. J. Soil Water Conserv. 57 (1): 29–37.

BESKOW, S. – MELLO, C.R. – NORTON, L.D. – CURI, N. – VIOLA, M.R. – AVANZI, J.C. (2009): Soil erosion prediction in the Grande River Basin, Brazil using distributed modeling. Catena 79: 49–59.

BUG, J. (2011): Modellierung der linearen Erosion und des Risikos von Partikeleinträgen in Gewässer. Entwicklung und Anwendung von entscheidungsbasierten Modellen zur flächenhaften Prognose der linearen Erosionsaktivitäten und des Gewässeranschlusses von Ackerflächen. Doctoral thesis. Institute of Physical Geography and Landscape Ecology, University of Hannover. 24–100.

CENTERI, CS. (2001): Az általános talajveszteség becslési egyenlet (USLE) K tényezőjének vizsgálata. [Investigation of the K factor of the Universal Soil Loss Equation.] Doctoral thesis. Szent István University, Gödöllő. 13–14, 44–63. (in Hungarian)

CHANG, M. (2006): Forest hydrology. Taylor and Francis, Boca Raton. 237–282.

CSÁFORDI, P. (2010): Erózióveszélyeztetettség vizsgálata a Soproni-hegység erdősült kisvízgyűjtőjén az USLE és az EROSION-3D modellekkel. [Soil erosion risk assessment with the models USLE and EROSION-3D in the small forested catchment of Sopron Hills.] In: Proceedings of the "Environmental management conference for the liveable countryside". Siófok, Hungary. September 2010. 189–198. (in Hungarian)

CSÁFORDI, P. – GRIBOVSZKI, Z. – KALICZ, P. (2010): Contribution of surface erosion to sediment transport in a small forested headwater catchment in the Sopron Hills. Journal of Landscape Management 1 (2): 3–11.

CSÁFORDI, P. – KALICZ, P. – GRIBOVSZKI, Z. (2011): Erdősült kisvízgyűjtő éves hordalékhozamának becslése és egy hordalékkúp hatásának vizsgálata. [Estimation of annual sediment load and the depletion effect of a sediment trap in a small forested catchment.] Hidrológiai Közlöny [Journal of the Hungarian Hydrological Society] 91 (3): 46–54. (in Hungarian)

DE ROO, A.P.J. – WESSELING, C.G. – RITSEMA, C.J. (1996): A single-event physically based hyrological and soil erosion model for drainage basins. I: Theory, input and output. Hydrol. Proc. 10: 1107–1117.

DE ROO, A.P.J. – JETTEN, V.G. (1999): Calibrating and validating the LISEM model for two data sets from the Netherlands and South Africa. Catena 37: 477–493.

DEMIRCI, A. – KARABURUN, A. (2011): Estimation of soil erosion using RUSLE in a GIS framework: a case study in the Buyukcekmece Lake watershed, northwest Turkey. Environ. Earth Sci., DOI 10.1007/s12665-011-1300-9

ERDOGAN, E.H. – ERPUL, G. – BAYRAMIN, I. (2007): Use of USLE/GIS Methodology for Predicting Soil Loss in a Semiarid Agricultural Watershed. Environ. Monit. Assess. 131: 153–161.

FISTIKOGLU, O. – HARMANCIOGLU, N.B. (2002): Integration of GIS with USLE in Assessment of Soil Erosion. Water Resour. Manage. 16: 447–467.

GARCÍA-RUIZ, J. M. – REGÜÉS, D. – ALVERA, B. – LANA-RENAULT, N. – SERRANO-MUELA, P. – NADAL-ROMERO, E. – NAVAS, A. – LATRON, J. – MARTÍ-BONO, C. – ARNÁEZ, J. (2008): Flood generation and sediment transport in experimental catchments affected by land use changes in the central Pyrenees. J. Hydrol. 356: 245–260.

GOMI, T. – MOORE, R.D. – HASSAN, M.A. (2005): Suspended sediment dynamics in small forest streams of the Pacific Northwest. JAWRA 41 (4): 877–898.

GORDON, N.D. – MCMAHON, T.A. – FINLAYSON, B.L. – GIPPEL, C.J. – NATHAN, R.J. (2004): Stream hydrology. John Wiley and Sons, Chichester. 169–200.

GRIBOVSZKI, Z. (2000): Forest stream sediment transport in Sopron Mountain. In: Tóth, L. (ed.): Hungarian Agricultural Engineering 13. Gödöllő. 78–80.

GRIBOVSZKI, Z. – KALICZ, P. (2003): Analysis of headwater stream sediment parameters. In: Geophysical Research Abstracts. European Geophysical Society, 28th Joint Assembly. Nice, France. April 2003.

JAIN, M.K. – KOTHYARI, U.C. (2000): Estimation of soil erosion and sediment yield using GIS. Hydrolog. Sci. J. 45 (5): 771–786.

JAIN, M.K. – MISHRA, S.K. – SHAH, R.B. (2010): Estimation of sediment yield and areas vulnerable to soil erosion and deposition in a Himalayan watershed using GIS. Curr. Sci. India 98 (2): 213–221.

KERTÉSZ, Á. – MÁRKUS, B. – MEZŐSI, G. (1992): Soil erosion assessment using GIS methods. In: 3rd European Conference on Geographical Information Systems. Munich, Germany. March 1992. 885–893.

KERTÉSZ, Á. – RICHTER, G. – SCHMIDT, R.G. (eds.) (1997): The Balaton project. In: European Society for Soil Conservation, Newsletter 2–3: 3–36.

KHOSROWPANAH, S. – HEITZ, L.F. – WEN, Y. – PARK, M. (2007): Developing a GIS-based soil erosion potential model of the UGUM watershed. Technical Report No. 117. Water and Environmental Research Institute of the Western Pacific, University of GUAM, UOG station, Mangilao, Guam. 9–33.

KUCSARA, M. – RÁCZ, J. (1988): A fajlagos évi eróziós talajveszteség vizsgálata a Tacsi-árok erdővel borított vízgyűjtő területén. [Examination of specific annual soil loss in the forested catchment of Tacsi Valley.] Erdészeti és Faipari Közlemények (1–2): 81–88.

LEE, S. (2004): Soil erosion assessment and its verification using the Universal Soil Loss Equation and Geographic Information System: a case study at Boun, Korea Environ. Geol. 45: 457–465.

LEWIS, J. (1998): Evaluating the impacts of logging activities on erosion and suspended sediment transport in the Caspar Creek watersheds. In: Proceedings of the "Conference on coastal watersheds: The Caspar Creek story". Ukiah, USA. 1998. 55–69.

LISLE, T.E. – NAPOLITANO, M.B. (1998): Effects of recent logging on the main channel of North Fork Caspar Creek. Technical report PSW GTR-168. Albany, CA, Pacific Southwest Research Station, Forest Service, USDA. 81–85.

LU, D. – LI, G. – VALLADARES, G.S. – BATISTELLA, M. (2004): Mapping soil erosion risk in Rondonia, Brazilian Amazonia: Using RUSLE, remote sensing and GIS. Land Degrad. Develop. 15: 499–512.

MA, J. (2001): Combining the USLE and GIS/ArcView for Soil Erosion Estimation in Fall Creek Watershed in Ithaca, New York. CSS620- Spatial Modeling and Analysis. Online: http://www.docstoc.com/docs/52400644/Combining-the-USLE-and-GISArcView-for-Soil-Erosion-Estimation (Access date: 02.12.2011)

MA, J.W. – XUE, Y. – MA, C.F. – WANG, Z.G. (2003): A data fusion approach for soil erosion monitoring in the Upper Yangtze River Basin of China based on Universal Soil Loss Equation (USLE) model. Int. J. Remote Sensing 24 (23): 4777–4789.

MARTIN, A. – GUNTER, J.T. – REGENS, J.L. (2003): Estimating erosion in a riverine watershed Bayou Liberty-Tchefuncta River in Louisiana. Environ. Sci. & Pollut. Res. 10 (4): 245–250.

MARKUS, B. – WOJTASZEK, M. (1993a): A talajerózió becslése [Soil erosion estimation]. Vízügyi Közlemények 75 (2): 192–200.

MÁRKUS, B. – WOJTASZEK, M. (1993b): Estimation of soil erosion using USLE and applying ARC/INFO and remote sensing. BME Newsletter, Budapest.

MOORE, I. – BURCH, G. (1986): Physical basis of the length-slope factor in the universal soil loss equation. Soil. Sci. Soc. Am. J. 50: 1294–1298.

MOSIMANN, T. – SANDERS, S. – BRUNOTTE, J. – GROCHOLL, J. – MARAHRENS, S. – PREUSS, E. – STREUN, M. (2004): Bodenerosion selber abschätzen. Ein Schlüssel für Betriebsleiter und Berater in Niedersachsen. Institute of Physical Geography and Landscape Ecology, University of Hannover. 3–29.

NEARING, M.A. – FOSTER, G.R. – LANE, L.J. – FINKNER, S.C. (1989): A process-based soil erosion model for USDA-Water Erosion Prediction Project technology. Transactions of the ASAE 32 (5): 1587–1593.

ONYANDO, J.O. – KISOYAN, P. – CHEMELIL, M.C. (2005): Estimation of Potential Soil Erosion for River Perkerra Catchment in Kenya. Water Resour. Manage. 19: 133–143.

PANDEY, A. – CHOWDARY, V.M. – MAL, B.C. (2007): Identification of critical erosion prone areas in the small agricultural watershed using USLE, GIS and remote sensing. Water Resour. Manage. 21: 729–746.

PFAFF, R.M. – GLENNON, J.A. (2004): Building a groundwater protection model. ArcUser July-September 2004: 54–59.

RÁCZ, J. (1985): Erdővel borított lejtős területek ellenállása az erózióval szemben. [Soil erosion resistance of forested sloping areas.] In: Proceedings of the conference "The economical management of sloping areas". Gödöllő, Hungary. 1985. 196–200. (in Hungarian)

RISSE, L.M. – NEARING, M.A. – NICKS, A.D. – LAFLEN, J.M. (1993): Error assessment in the Universal Soil Loss Equation. SSSA 57 (3): 825–833.

RODDA, H.J.E. – DEMUTH, S. – SHANKAR, U. (1999): The application of a GIS-based decision support system to predict nitrate leaching to groundwater in southern Germany. Hydrolog. Sci. J. 44 (2): 221–236.

SANDERS, S. (2007): Erosionsmindernde Wirkung von Intervallbegrünungen in Fahrgassen. Untersuchungen im Weizen- und Zuckerrübenanbau mit Folgerungen für Anbaupraxis. Series: Geosynthesis. Institute of Physical Geography and Landscape Ecology, University of Hannover.

SCHWERTMANN, U. – VOGL, W. – KAINZ, M. – AUERSWALD, K. – MARTIN, W. (1987): Bodenerosion durch Wasser: Vorhersage des Abtrags und Bewertung von Gegenmaßnahmen. Ulmer, Stuttgart. 9–36.

SHEN, H.W. – JULIEN, P.Y. (1993): Erosion and sediment transport. In: Maidment, D.R. (ed.): Handbook of hydrology. McGraw-Hill. 12.1–12.61.

SORRISO-VALVO, M. – BRYAN, R.B. – YAIR, A. – IOVINO, F. – ANTRONICO, L. (1995): Impact of afforestation on hydrological response and sediment production in small Calabrian catchment. Catena 25: 89–104.

STONE, R.P. – HILBORN, D. (2000): Universal Soil Loss Equation (USLE). Series: Factsheet No. 00-001 AGDEX 572/751. Agricultural Engineering, Ministry of Agriculture, Food and Rural Affairs, Ontario. May 2000.

SURFLEET, C.G. – ZIEMER, R.R. (1996): Effects of forest harvesting on large organic debris in coastal streams. In: Proceedings of the "Conference on coast redwood forest ecology and management". Arcata, CA, Berkeley, CA, University of California. June 1996. 134–136.

ÚJVÁRI, F. (1981): Az erdők szerepének értékelése a vízgyűjtő területek hordalék-lemosódásának megakadályozásában. [Assessment of the soil protection role of forests in the catchment areas.] Erdészeti Kutatások 74: 107–124. (in Hungarian)

US EPA (2009): Water models / Barton Springs Salamander – Meadow. US Environmental Protection Agency. Online: http://www.epa.gov/oppefed1/models/water/bs_meadow.htm (Access date: 30.05.2010)

VAN REMORTEL, R.D. – MAICHLE, R.W. – HICKEY, R.J. (2004): Computing the LS factor for the Revised Universal Soil Loss Equation through array-based slope processing of digital elevation data using a C + + executable. Comput. Geosci. 30: 1043–1053.

VON WERNER (1995): GIS-orientierte Methoden der digitalen Reliefanalyse zur Modellierung von Bodenerosion in kleinen Einzugsgebieten. Doctoral thesis. Freie Universität Berlin.

WISCHMEIER, W.H. – SMITH, D.D. (1978): Predicting rainfall erosion losses – A guide to conservation planning. Series: Agriculture Handbook No. 537. USDA, Washington DC. 3–4.

VERSITA DOI: 10.2478/v10303-012-0005-4 Acta Silv. Lign. Hung., Vol. 8 (2012) 57–74

# Climate Change and Conservation

F. Thomas LEDIG*

Department of Plant Sciences, University of California, Davis, California

**Abstract** – Conserving forest genetic resources and, indeed, preventing species extinctions will be complicated by the expected changes in climate projected for the next century and beyond. This paper uses case examples from rare spruces (*Picea* sp.) from North America to discuss the interplay of conservation, genetics, and climate change. New models show how climate change will affect these spruces, making it necessary to relocate them if they are to survive, a tool known as assisted migration or, preferably, assisted colonization. The paper concludes with some speculation on the broader implications of climate change, and the relevance of conservation to preserving the necessary ecological services provided by forests.

**conservation genetics / assisted colonization / extinction / *Picea mexicana* / *Picea martinezii* / *Picea chihuahuana* / *Picea breweriana***

**Kivonat – Klímaváltozás és génmegőrzés.** Az erdészeti genetikai erőforrások megőrzése, és különösen a fajkihalások megakadályozása az évszázad végére (és azutánra) előrevetített klímaváltozás fényében nehéz feladatnak tűnik. A tanulmány ritka észak-amerikai lucfenyő fajok példáján tárgyalja a klímaváltozásból adódó genetikai és megőrzési problémák együttesét. Modellek mutatják be klímaváltozás hatását a luc fajok elterjedésére. Ezek szerint szükség lesz a fajok megmentése érdekében a mesterséges áttelepítés, ill. kolonizáció megoldására. A tanulmány kitér a klímaváltozás tágabb következményeire és a konzervációnak az erdőtakaró ökológiai szolgáltatásaiban játszott szerepére.

**konzerváció-genetika / mesterséges kolonizáció / kihalás / *Picea mexicana* / *Picea martinezii* / *Picea chihuahuana* / *Picea breweriana***

## 1 INTRODUCTION: EX SITU AND IN SITU CONSERVATION

Conservation efforts can be placed into two categories: ex situ, meaning out of place, such as in clone and seed banks, and in situ, meaning in place, as in natural reserves. In situ conservation to most people meant natural forests in protected areas. In the last several decades, most advocates for the conservation of forest genetic resources in North America concluded that in situ conservation had significant advantages over ex situ conservation.

* tomledig100@gmail.com; University of California, Davis 95616 USA

Both ex situ and in situ conservation can provide for future economic values, such as the discovery of new products, like taxol (Wheeler – Hehnen 1993). But, in situ conservation has other values, such as ecologic: forests provide services through the carbon cycle and many other geochemical and hydrologic cycles. Esthetic values: forests enrich our world, both by their own beauty and because of the many creatures they shelter which, by the way, it may be our moral duty to protect, according to most of the world's major religions. In situ conservation does all this with a minimum of input and expense.

However, in the future, probably no temperate forest will be truly natural and wild, even in North America. They will not be natural because society will have to move species and forests to prevent extinctions due to rapid climate change combined with limited seed dispersal. Although forest species and genetic resources cannot be maintained solely in seed banks, nevertheless, they will not be maintained without seed banks either, or at least without some form of ex situ conservation. Assisted colonization will be necessary to prevent the loss of genetic diversity and even species extinctions, but assisted colonization (or reforestation or some other term), also will be necessary to restore forests and the ecosystem services that they provide as current forests are reduced by climate change.

### 1.1 Gene Conservation and Conservation Genetics

To facilitate discussion of forest conservation, some terminology needs to be clarified; in particular, the phrases 'gene conservation' versus 'conservation genetics'. Gene conservation is about the preservation of genes for their future utility to breeders. That can be accomplished ex situ, in bottles and bags – seed banks.

Conservation genetics is about the genetic conditions for survival, reproduction, and evolution in response to changing environments. This is about maintaining healthy forests, which cannot be done in bottles. However, genetically healthy forests serve the purposes of gene conservation just as well or better than seed banks. Conservationists have talked far too much about gene conservation and far too little about conservation genetics.

### 1.2 Genetics and Extinction

External pressures threaten the survival of species, populations, and genetic resources, and therefore, threaten the forests themselves. These threats are usually human-caused. Among these external pressures, the most important may be habitat destruction, exploitation, introduced pests and diseases, pollution, and climate change. What exactly do these threats do? It is so simple, it is almost trivial. Their effect is to fragment large populations and reduce them in size. When populations are small enough, they are classed as threatened or endangered.

Whatever the external factors (development, over-harvest, etc.) that begin the march toward extinction, genetic processes end it. The pioneering conservation biologists, Michael Gilpin and Michael Soulé (1986), developed an analogy to a whirlpool or vortex (*Figure 1*). The external threats reduce population size and fragment larger, more extensive populations into smaller units. Small populations lose genetic diversity as a result of random genetic drift. Loss of genetic diversity limits adaptation to changing conditions, which leads to even smaller populations. The smaller population, the more inbreeding, even if mating is completely at random, because the choice of mates is reduced. Inbreeding results in loss of fitness in most organisms, and the population becomes even smaller. Fragmentation increases separation among subpopulations, which means less gene flow within the metapopulation, and genetic diversity can no longer be maintained, contributing to the reduction in size. Demographic variation becomes more likely in small populations, ratcheting them still lower in size. And so on in a feedback

circle, until these stochastic processes drive the population to extinction.

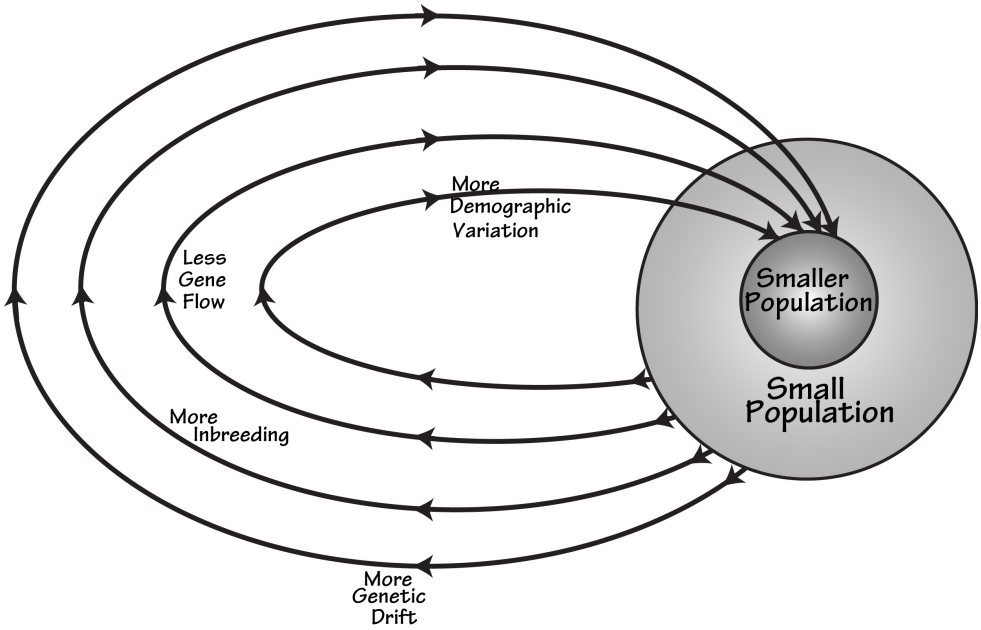



*Figure 1. The genetic extinction vortex (Gilpin – Soulé 1986). External threats – urban and agricultural development, climate change, invasive species and diseases, overexploitation, and other factors – reduce available habitat and fragment species into smaller population. The smaller the population, the more genetic drift, which reduces diversity. Diversity cannot be replaced by gene flow because the fragments are isolated by distance The reduction in genetic diversity limits adaptation to environmental changes. The smaller the population, the more inbreeding and inbreeding reduces fitness. All of these stochastic genetic processes operate along with demographic stochasticity and tend to further reduce population size in a positive feedback cycle until extinction is inevitable. All extinctions, with the exception of catastrophic extinction, may be the result of genetic factors in the final analysis.*

This is documented in the case of the greater prairie chicken (*Tympanuchus cupido*) where even habitat improvement could no longer save the subpopulation in Illinois, USA (Westemeier et al. 1988), and in the Glanville fritillary (*Melitaea cinxia*), the first report of extinction linked to lack of genetic diversity and inbreeding in natural populations in the Aland Islands (Saccheri et al. 1998).

### 1.3 Genetics in Conservation

If genetics is the final factor in the destruction of forest species, genetic knowledge can also help in conservation. Some of the ways are: clarifying taxonomy; guiding the choice of where to place reserves; and diagnosing problems, such as lack of diversity, lack of gene flow, or excessive inbreeding.

The most basic way genetics enters into the conservation decision is through taxonomy. We must know what is out in the forest before we can even think about ways to save it. What should be saved? For example, *Picea mexicana* and *Picea martinezii* were not recognized as species eight years ago. The foremost authority on conifers (Farjon 2001) treated *Picea mexicana* as just a few, isolated, Mexican populations of a more common northern species, *Picea engelmannii*; and *Picea martinezii* was considered identical to *Picea chihuahuana*.

Without taxonomic status, there can be no protection, neither under law nor through public awareness.

We now know from genetic studies using markers, both isozymes and from DNA, that *Picea mexicana* is very different from *Picea engelmannii* (*Figure 2*). The data indicate that *Picea martinezii* and *Picea chihuahuana* have been separate species for 2 to 3 million years (Ledig et al. 2004).

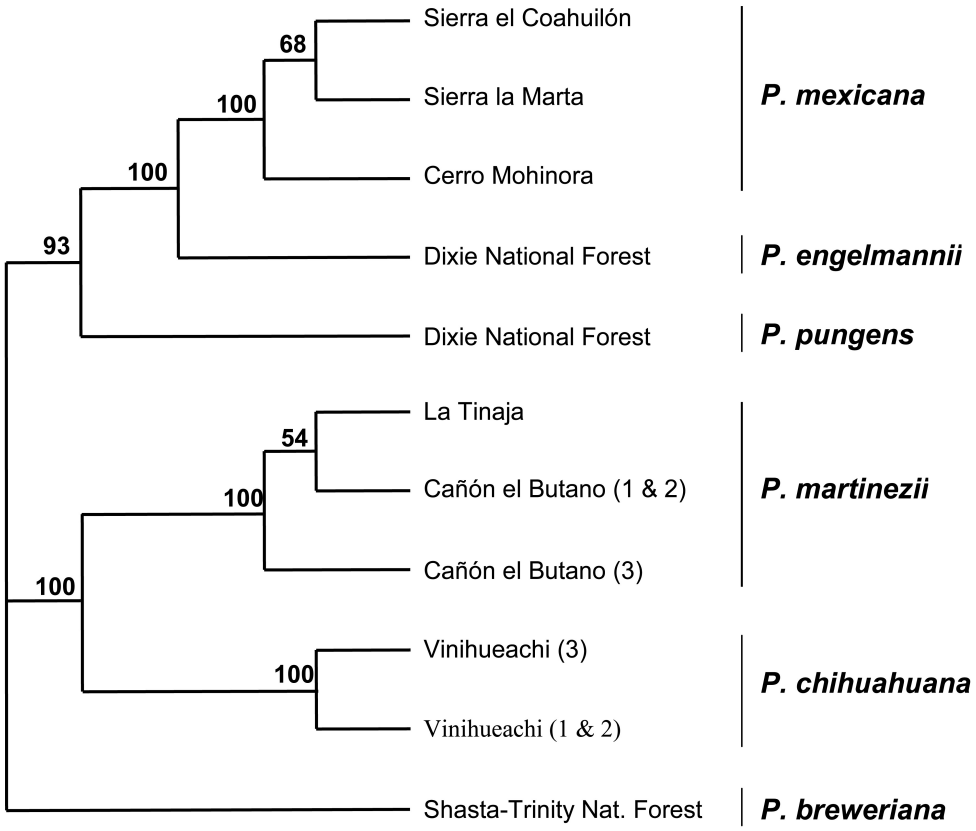



*Figure 2. Neighbor-joining phylogenetic tree based on dissimilarity distance (Nei – Li 1985)*

*Figure 2* demonstrates that *Picea chihuahuana*, *P. martinezii*, and *P. mexicana* are separate species and that they are distinct from *P. engelmannii*. *P. pungens* is sometimes confused with *P. engelmannii* but the two are obviously distinct. *P. breweriana* was used as an outgroup to anchor the tree. Numbers near nodes are percentage of 1000 bootstrap replicates that justify the node (Ledig et al. 2004).

## 2. RESULT: CASE EXAMPLES – MEXICAN SPRUCES

The Mexican spruces are largely unknown to most foresters, even in Mexico. There are three species of spruce in México (*Figure 3a,b,c*). They have highly fragmented ranges, small populations, and are relicts of past glacial periods. Localized, fragmented species are always more at risk of extinction than widespread species, such as Canada's transcontinental spruces.

*3a: Picea mexicana:*
*3 populations spread over 676 km,*
*tallest peaks – 3350–3550 m, < 4,000 trees*

*3b: Picea chihuahuana:*
*39 populations spread over 687 km,*
*ravines at 2155–2990 m,*
*24,211 trees > 2 m tall*

*3c: Picea martinezii:*
*6 populations spread over 147 km,*
*at elevations 1820–2515 m < 2,000 trees*

*Figure 3. Photos and some data on the rare, little-known Mexican spruces, which occur in remote areas of the Sierra Madre Oriental and Sierra Madre Occidental*

Studies of the Mexican spruces began because of the North American Forest Commission and its Forest Genetic Resources Working Group (FGRWG), which includes three delegates from each country in North America – Canada, México, and the US. *Picea chihuahuana* first drew our attention. However, all the Mexican spruces are wonderful models to study the effects of fragmentation and population size on genetic processes. The FGRWG undertook studies of the Mexican spruces in a spirit of international cooperation.

A great deal of international cooperation was needed. Many colleagues and I worked together and as a result learned a lot about these rare spruces. The project was begun largely thanks to Teobaldo Eguiluz, the founding director of the Centro de Genética Forestal in Chapingo. My late colleague, Basilio Bermejo, shouldered much of the burden when he became director of the Centro de Genética Forestal. However, Celestino Flores deserves most of the credit for carrying out extensive investigations in the mountains of Mexico.

*Picea mexicana* is the rarest of the Mexican spruces (Ledig et al. 2000). It occurs in only three locations. Two locations are in the Sierra Madre Oriental, the eastern mountain range of Mexico. One location is on Sierra la Marta, where most of the population was destroyed by fire 30 years ago. Another population is on Sierra el Coahuilon, only 5 km from Sierra la Marta. These two peaks are among the highest in the Sierra Madre Oriental. The third location is on Cerro Mohinora, 676 km away in the Sierra Madre Occidental, the western mountain range of Mexico. Cerro Mohinora is the highest peak in the State of Chihuahua.

Like *Picea mexicana*, *Picea martinezii* also occurs in the Sierra Madre Oriental. When I began work on the Mexican spruces, we knew of only two populations. These were discovered in 1984 by Glafiro Alanís and a visiting German professor, Burkhardt Müller-Using, at the Universidad Autónoma Nuevo León in Linares (Müller-Using – Alanís 1984, Müller-Using – Lässig 1986). One location was at a site called La Trinidad, a population of about 350 trees at the foot of El Butano, a cliff that rises over 200 m above the spruce. Another, separated from the first by 147 km, had about 68 trees. Both sites are in the State of Nuevo León. Later, Celestino Flores discovered four other groups of trees near Aramberri, a village in the middle of the desert at the base of a high sierra that provides refuge for remnants of the North American Arcto-Tertiary forest (Ledig et al. 2000).

*Picea chihuahuana* is more widespread than the other two Mexican spruces. We have found 39 populations (Ledig et al. 2000). They are spaced over 687 km north-south in the Sierra Madre Occidental – north and south of the famous Copper Canyon country, Cañon del Cobre. Chihuahua spruce almost always occurs in steep arroyos where the sun may not reach until noon.

The map (*Figure 4*) gives some idea of how fragmented these three species are. All of these species are considered endangered because of the small numbers of populations and the small numbers of trees. Manuel Mápula and Celestino Flores counted and measured every tree of *Picea chihuahuana* and all saplings and seedlings above 0.3 m tall (Ledig et al. 2000). There are less than 25,000 trees in the 39 stands.

Data on the other two species are not precise, but *Picea martinezii* certainly has fewer than 2000 trees, probably less than 1000. *Picea mexicana* has only three stands and probably less than 4000 trees.

There are other demographic reasons to be worried about these species. Reproduction is scarce. For *Picea chihuahuana*, on which we have good data, the ratio of seedlings and saplings to trees is less than in northern spruces, such as *Picea rubens* and *Picea engelmannii* (*Table 1*).

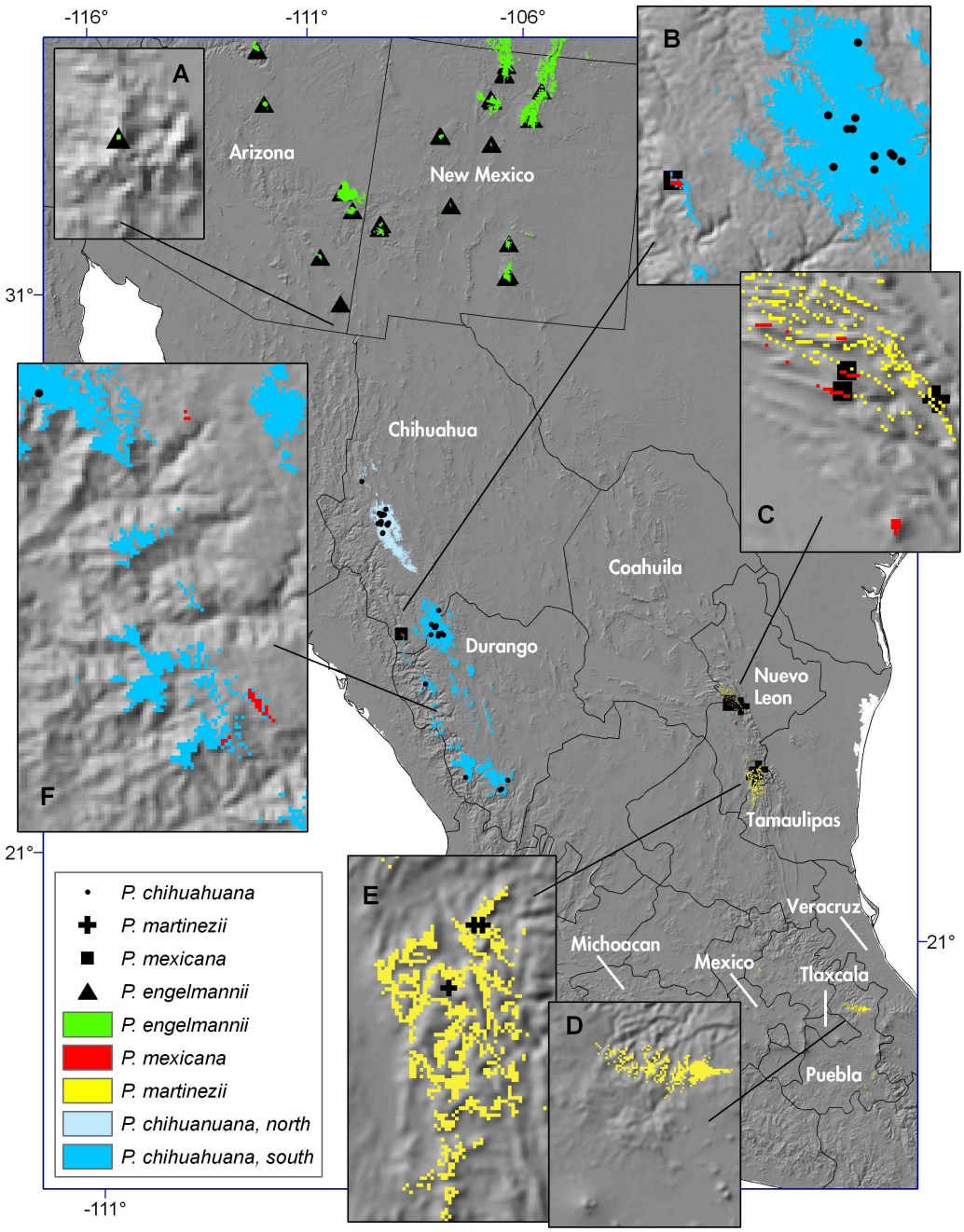



*Figure 4. Map of Mexico and the extreme southwestern United States showing all the present locations of Picea chihuahuana (●), P. martinezii (□), P. mexicana (■), and for comparison, locations of P. engelmannii (▲) in the southwestern United States. The areas predicted to be climatically suitable for the two mitotypes of P. chihuahuana are in light blue (northern mitotype) and dark blue (southern mitotype). The yellow is where the climate is predicted suitable for P. martinezii and the red is predicted to be climatically suitable for P. mexicana. The insets magnify detail in significant areas. See Ledig et al. (2010) for details on the modeling procedure.*

*Table 1. The ratio of seedlings and saplings to trees ( >2 m height) in Picea chihuahuana, P. rubens, and P. engelmannii. Data for the latter two species are from Burns and Honkala (1990).*

| Species | Mean | Range |
|---|---|---|
| *Picea chihuahuana* | 0.59 (±0.05) | 0.09 – 1.47[1] |
| *P. rubens* | 13.02 | 1.87 – 26.85 |
| *P. engelmannii* | 2.34 | 0.24 – 7.33 |

[1] *Omitting an outlier of 3.18, a stand which had been recently harvested, leaving predominantly seedlings and saplings.*

Why is reproduction scarce? Seed germination may be an explanation, but not the ultimate explanation. The highest value we obtained for seed germination was 54% in seeds from one of the largest stands of *Picea chihuahuana*, one which has over 2600 trees and is only a few kilometers from a stand with over 3500 trees (Ledig, unpublished data); in other words, a large, healthy population. Seeds from other, smaller populations had lower rates, approaching zero in many small stands. In the two *Picea martinezii* stands of about 60 and 350 trees, germination averaged 7%. Much older seeds of the widespread U.S. species, *Picea engelmannii* (about 20 years old when tested) had substantially higher germination rates, about 85%. Therefore, some of the reason for the low seedling-sapling:tree ratio of *Picea chihuahuana* might be a result of low seed germinabilty.

### 2.1 Conservation Genetics and the Mexican spruces

We used molecular markers, mostly isozymes, to analyze genetic variation in all three spruces (Ledig et al. 1997, 2000, 2002), providing extensive knowledge of their population genetic structure. I've mentioned above that the data showed convincingly that the Mexican spruces are phylogenetically distinct from each other and from the US *Picea engelmannii* (Ledig et al. 2004).

We also know that genetic variation is lower than normal for conifers. What does this tell managers? That they should concentrate protection on the populations with the greatest genetic variation because those populations have the greatest potential for adaptive evolution.

We know that today chance – random genetic drift – is the most important genetic process in these populations, overwhelming natural selection. In these situations, mutations reach high frequency even as the population loses genetic variation. And these mutations reduce fitness because the vast majority are deleterious. What does this tell managers? Managers should help the populations increase in size through silvicultural methods and enrichment plantings so that genetic drift is less important.

We know that exchange of genes among populations by pollen or by seed dispersal is near zero, especially, in *Picea chihuahuana*. Therefore, much of the genetic variation is among populations. What does this tell managers? Managers should encourage gene flow and maintain as many populations as possible.

We have looked at the mating system – the proportion of self-fertilization and outcrossing – and found that selfing is very high, sometimes 100%. This is completely unique in conifers. The result is inbreeding and a great loss in fitness. This inbreeding is the cause of low seed yields and low germination rates, which explains why regeneration is scarce. What does this tell managers? Managers should encourage outcrossing.

There is a common thread to these management suggestions. If we encouraged gene flow among populations, it would increase their genetic variation and reduce inbreeding and reduce genetic drift. There are several ways to do this. We could connect populations with migration corridors to promote gene flow. However, the stands are separated by great distances and intervening habitat is likely to be unsuitable for these species; e.g., *Picea mexicana* is restricted to only the highest peaks in the Sierra Madre Oriental and the Sierra Madre Occidental and these ranges are very distant from one another. No migration corridor is possible. Another method would be to pollinate one population with pollen from another. However, anyone who has cross-pollinated trees knows that this is wildly impractical. The most practical way of encouraging gene flow and increasing genetic diversity is by enrichment plantings. That is, collect seeds from various populations, germinate the seeds and grow seedlings from them in a forest nursery, and, finally, interplant the seedlings among different stands. When these grow to tree size, they should naturally intercross with the locals, restoring genetic diversity and eliminating the effects of genetic drift and inbreeding.

### 2.2 Climate Change and the Mexican Spruces

The management suggestions outlined above would be sufficient to restore genetic health if the Mexican spruces existed in a static world. However, the world is not static. Change, climate change in the past, is the reason the Mexican spruces have highly fragmented ranges and small populations. During the last glacial maximum, spruce in México extended at least 700 km further south to the Basin of México (Clisby – Sears 1955, Lozano-García et al. 1983). Today, after Holocene warming, they are mere relicts.

Foresters and conservationists must consider the effects of climate change projected for this century and beyond (Mátyás 2006). Assisted colonization – relocation – will be necessary to conserve genetic resources and prevent extinctions. Assisted colonization has been hotly debated in the last decade (see the discussion in Ledig et al. 2012), and I will not rehash that here. Assuming that assisted colonization is accepted, our first guess might be that species must be moved up in elevation or northward. However, *Picea mexicana*, for example, is already at the very top of the tallest peaks in the Sierra Madre Oriental and Sierra Madre Occidental. Where could it go? To answer that question, one must know the future distribution of climatic niches.

For projections of the effect of climate change, I am especially indebted to Jerry Rehfeldt and Cuauhtémoc Sáenz. Cuauhtémoc Sáenz developed spline climatic models for all of México (Sáenz-Romero et al. 2010) and Jerry Rehfeldt has calculated and modeled the future climate for all these rare spruces (Ledig et al. 2010).

We modeled the present fundamental climatic niches for all three Mexican spruces and for southwestern populations of *Picea engelmannii* (*Figure 4*). For this purpose, *Picea chihuahuana* was divided into two genetically different groups – north and south – called mitotypes because they differ in their mitochondrial DNA (Jaramillo-Correa et al. 2006).

The model correctly predicts where *Picea mexicana* and *Picea martinezii* now grow in the northern Mexican States of Coahuila and Nuevo León. However, it also shows that many other sites are climatically suitable, such as a peak called Cerro Potosí in southern Nuevo León predicted suitable for *Picea mexicana*. The reason these species do not grow everywhere the climate is suitable is because these projections show the fundamental climatic niche, not the realized niche. The realized niche may be affected, in part, by soils or by interactions with other species – competitors, insect pests, and pathogens.

The model correctly predicts *Picea mexicana* on Cerro Mohinora in the western Mexican State of Chihuahua, but indicates that conditions lower on the mountain might be right for

*Picea chihuahuana* as well. However, it also shows suitable habitat for *Picea mexicana* southward on some high ridges in the State of Durango, where it does not now occur.

For *Picea martinezii*, the model predicts suitable habitat in many areas where it is not actually found in the northern States of Mexico, Coahuila and Nuevo León and on the border of the States of Nuevo León and Tamaulipas. Note that the present climate far south in the northern part of the State of Puebla in central Mexico is also predicted to be suitable for *Picea martinezii*, but it does not occur there.

We also projected the situation for 2060 (*Figure 5*) using Global or General Climate Models (GCMs). GCMs predict future climates. By 2060, the suitable habitat for each species shrinks. In fact, the fundamental niche for *Picea chihuahuana* disappears in Chihuahua. The model shows both mitotypes overlapping near the Chihuahua-Durango border – the northern mitotype on the eastern side and the southern mitotype on the western side of the Sierra Madre Occidental. On Cerro Mohinora, *Picea chihuahuana* replaces *Picea mexicana*.

*Picea mexicana* also disappears near the Coahuila-Nuevo León border on Sierra la Marta and Sierra el Coahuilon. In fact, in 2060, its habitat is gone – completely! There is no suitable habitat for *Picea mexicana* in just 50 years.

For *Picea martinezii*, the climate niche shrinks in the States of Nuevo León, Coahuila, and Tamaulipas. However, it expands to the south in the east-central Mexican States of Veracruz, Puebla, and elsewhere in the Trans-Mexican Volcanic Belt (a belt of active vulcanism and high volcanic peaks that runs east-west across central Mexico). Peaks in the Volcanic Belt can reach 5636 m in elevation compared to only about 3500 in the Sierra Madre Oriental. The climate niche for *Picea martinezii* also appears for the first time in the State of México and in eastern Michoacán, both in central Mexico.

Notice that habitat does not open up in the north of México nor in the southwestern US for any of the three spruces.

By 2090, the climate for *Picea chihuahuana* is predicted to shrink even more drastically to a small area in the border region of the States of Chihuahua and Durango (Ledig et al. 2010). The climate suitable for *Picea martinezii* is almost gone in the area near the border of the northern States of Coahuila and Nuevo León, and has shrunk badly near the border of the States of Nuevo León and Tamaulipas. However, a climate niche for *Picea martinezii* still is predicted around the volcanic peak, Orizaba, the highest peak in Mexico, in other mountains along the Puebla-Veracruz border, and on the eastern border of the central Mexican State of Michoacán.

Furthermore, in 2090 a climatic niche is predicted for *Picea mexicana* for the first time on the volcanic peaks Tláloc, Iztaccíhuatl, Popocatépetl, and La Malinche in the Trans-Mexican Volcanic Belt (Ledig et al. 2010). Suitable climate for *Picea martinezii* is predicted below *Picea mexicana* on La Malinche.

In conclusion, the situation in the northern States of Mexico (Coahuila, Nuevo León, and Tamaulipas) is grim. My suggestion to Mexican foresters has been to start test planting *Picea martinezii* and *Picea mexicana* on the high volcanic peaks now. If seed banks are not established for these species soon, it may be too late.

*Picea chihuahuana* is an even bigger problem because the models do not show a suitable climatic niche developing, even in the Trans-Mexican Volcanic Belt. Where can it find suitable habitat? From the projections, that is an unsolved question. Perhaps, no suitable climate will develop in Mexico.

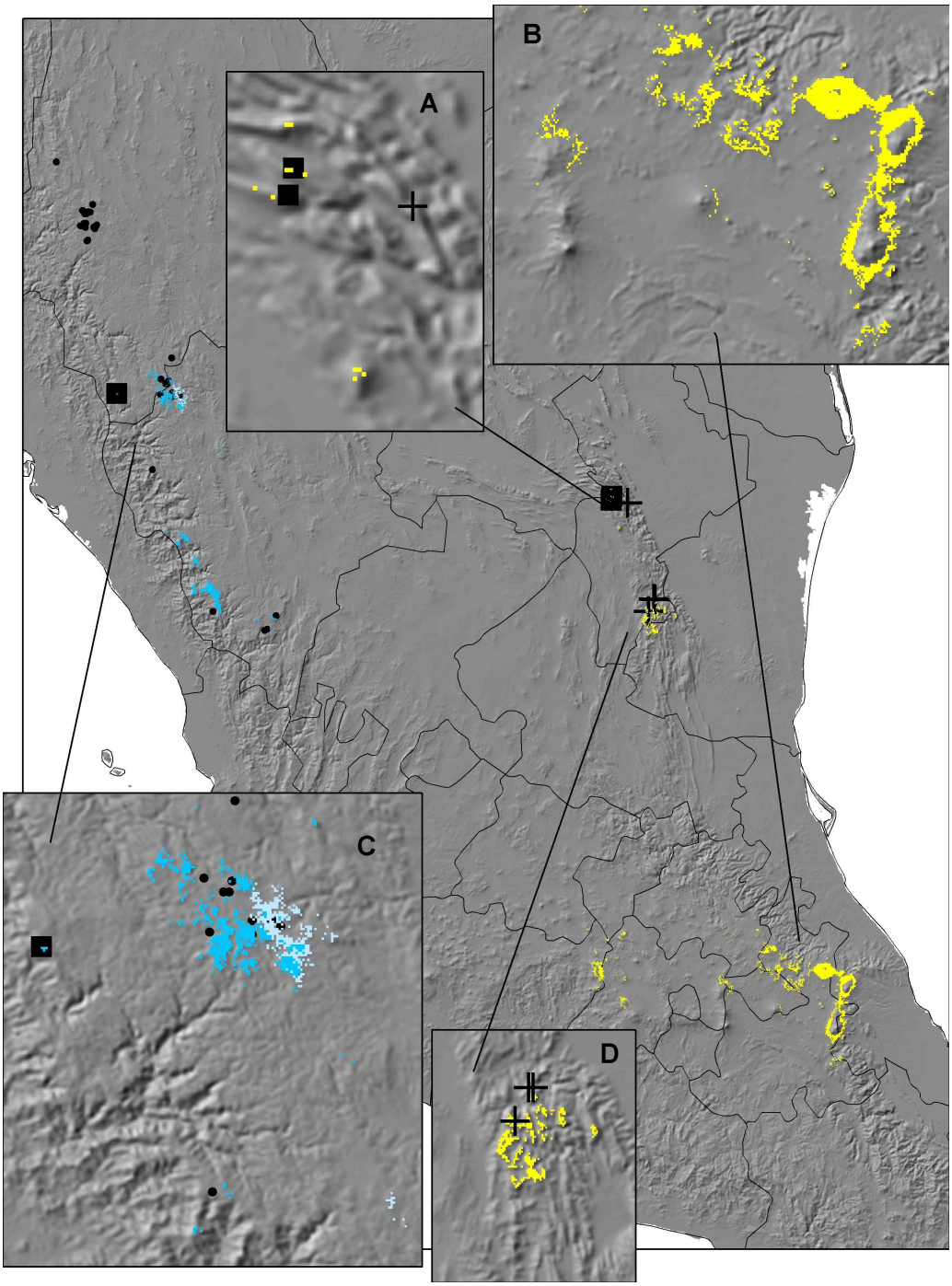



*Figure 5. Composite mapping of six projections made from output of three General Circulation Models (GCMs) and two scenarios (pessimistic – business as usual – and optimistic – some restraints on carbon emissions) for the decade surrounding 2060 using the majority of votes cast by the Random Forests classification tree to determine presence or absence of four taxa. Light blue for the northern mitotype of P. chihuahuana and sky blue for the southern mitotype; yellow for P. martinezii; and red for P. Mexicana. The insets magnify detail in significant areas. See Ledig et al. (2010) for details on the modeling procedure.*

## 3 CASE EXAMPLE – *PICEA BREWERIANA*

*Picea breweriana* is my last example. *Picea breweriana* is native only to the Klamath-Siskiyou Mountains of the West Coast of the United States – in the States of California and Oregon. The range is fragmented (*Figure 6*). According to fossil evidence, *Picea breweriana* was once a part of the Arcto-Tertiary forest (Wolfe 1964, Axelrod 1987). The Arcto-Tertiary forest spanned western North America.

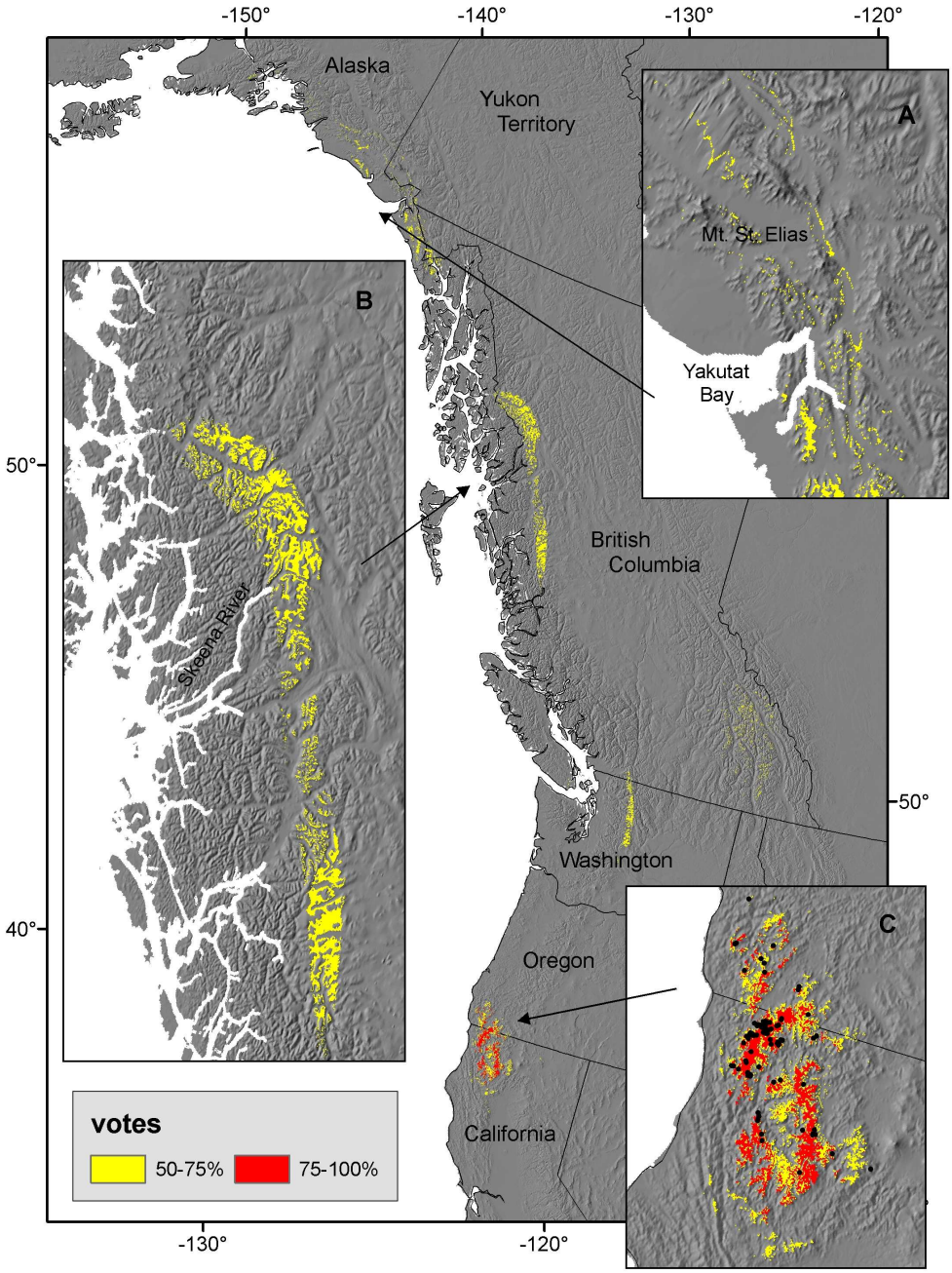



*Figure 6. Map of northwestern coastal North America showing the range of Picea breweriana (●) and areas of suitable climate as modeled by the Random Forests classification tree. Yellow means the climate was deemed suitable by one General Climate Model (GCM), blue by two GCMs, and red means the climate was deemed suitable by all three GCMs used in the modeling approach. See Ledig et al. (2012) for details on the modeling procedure.*

*Picea breweriana* has moderate levels of genetic diversity for a conifer, except in its northernmost outlying population in Oregon (Ledig et al. 2005). It is highly outcrossing – on average 92%, despite low stand density in several of our samples. Even when highly isolated trees are compared to those in dense clusters, there is no increase in the level of selfing. But, there is fairly high differentiation and low gene flow among populations. The high outcrossing rate is in striking contrast to that in the equally fragmented Mexican spruces.

The genetic analysis is mostly good news with regard to *Picea breweriana*. That is, genetic diversity is moderate – not high, but not low either. And the rate of outcrossing is high, which is very promising. The bad news is that at least some populations may have suffered loss of genetic variation because of genetic drift, and gene exchange among populations is low, which tends to isolate them, particularly those in Oregon. However, compared to the Mexican spruces, genetic analysis suggests that it should be quite simple to maintain viable populations of *Picea breweriana* in situ. Nevertheless, because of the high level of differentiation, I would again suggest that managers encourage gene flow, probably by exchanging seedlings among stands.

But what about climate change? Certainly, *Picea breweriana*, like the Mexican spruces, responded to past climate changes. *Picea breweriana* occurred all over the western United States 10 million years ago, before the climate became more arid in the post-Miocene. Its present habitat is a small fraction of its former range.

As he did for the Mexican spruces, Jerry Rehfeldt also modeled the fundamental climate niche of *Picea breweriana*; *Figure 6* includes its predicted present climatic niche (Ledig et al. 2012).

*Figure 7* is constructed with Global Climate Models (GCM) for the decade 2060. By 2060, there will be almost no suitable climatic niche in the lower United States for *Picea breweriana*. Only one GCM has it clinging to a few ridgetops. By 2090 (not shown; see Ledig et al. 2012) even these small areas of projected habitat are predicted to disappear.

Conversely, climate in northern British Columbia, the Yukon Territory, and southern coastal Alaska may be suitable for *Picea breweriana*. In *Figure 7*, blue means the climate was deemed suitable by two GCMs. And if the map could be seen in magnification, it would reveal a few red pixels; i.e., habitat suitable according to three GCMs. Note that *Picea breweriana* does seem to match our first notions of where to move species – north – in contrast to the situation in the Mexican spruces. The lesson is that every species and climate combination could be unique.

Without assisted colonization, *Picea breweriana*, like the three species of Mexican spruces, may go extinct. Therefore, Barry Jaquish (British Columbia Ministry of Forests, Lands and Resource Operations, personal communications, 2012) is planning provenance plantings in the Prince Rupert area of British Columbia, using seed collections from 10 populations. *Picea breweriana* is an example in which international cooperation may actually save a species from extinction.

### 3.1 Extinction, Genetics, Climate Change, and Assisted Colonization

To summarize to this point: A massive wave of extinction will be upon us in a short time because of the many unique threats resulting from human activities (Pimm et al. 1995, Ward 1995). Both now and later we should work to maintain genetically healthy populations, because beyond some point, small populations will find themselves in the genetic vortex from which they cannot escape (Gilpin – Soulé 1986). Saving genetic resources in bottles is not the final answer, but it may be a necessary interim tool because many climate models suggest such extensive rearrangements (Williams – Jackson 2007) that assisted colonization will be absolutely necessary to prevent extinctions. And assisted colonization requires some mixture of in situ and ex situ conservation methods.

I began my career 50 years ago by advocating conservation in situ in natural forests. But, if forests are moved to exotic locales, this is, in fact, ex situ conservation? In fact, forests of the future may have no present analogue.

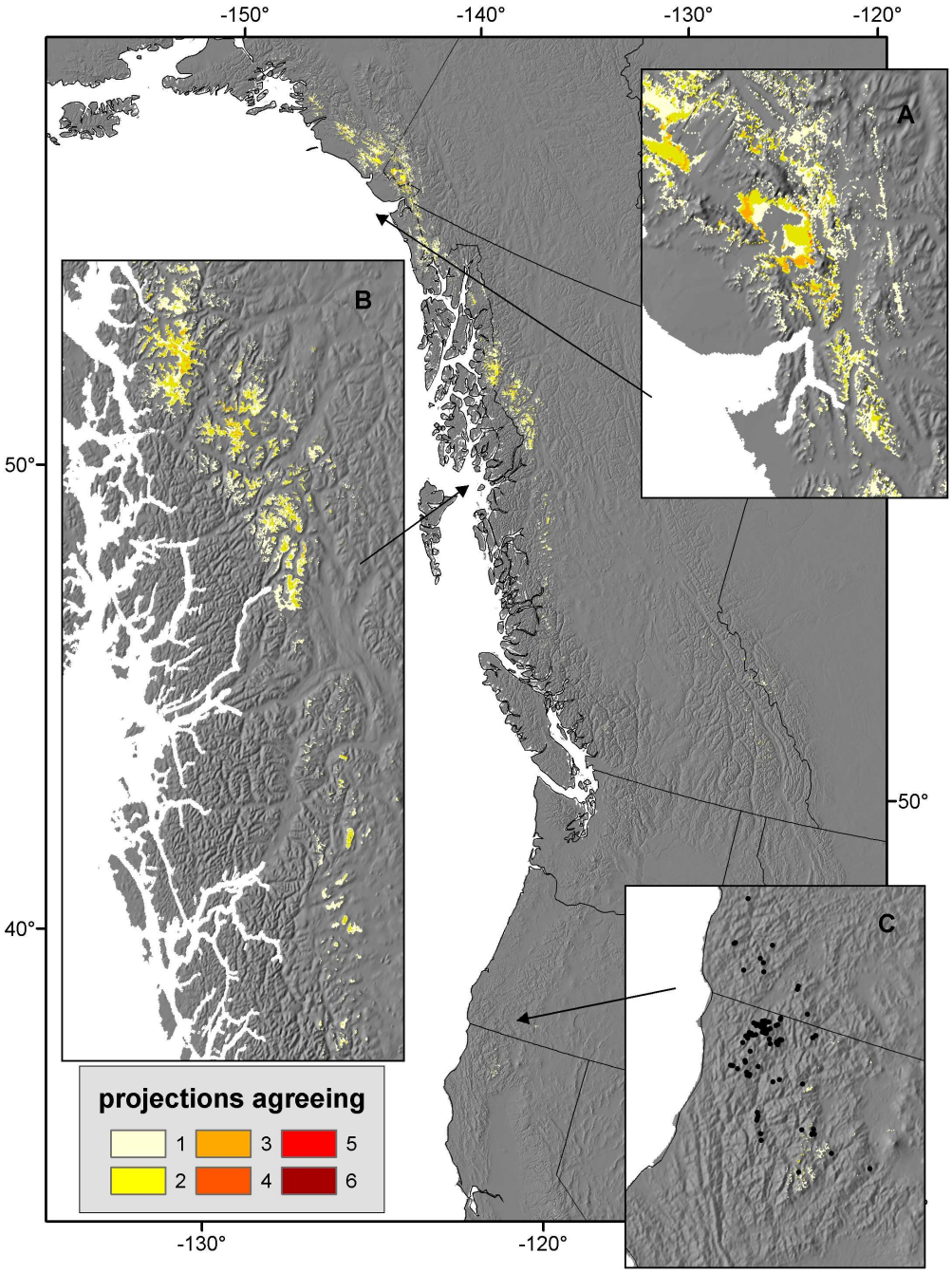



*Figure 7. Composite mapping of the climatic niche of Picea breweriana made from the Random Forests bioclimate model for six projections using output of three GCMs and two scenarios (pessimistic – business as usual – and optimistic – some restraints on carbon emissions) for the decade surrounding 2090. Climate suitability uses the 50 % voting threshold. Insets show in detail the Yakutat Bay - Mt. St. Elias region of Alaska (A), the Nass-Skeena region of British Columbia (B), and the Klamath region of California and Oregon (C). Dots in C locate sites inhabited by Picea breweriana that were used for the present analysis. Shades of color code concurrence: lightest yellow, 1 projection; darkest red, 6 projections.*

## 4 CONCLUSION

About 20 years ago, I wrote this (Ledig – Kitzmiller, 1992):

> *"Foresters ... must be ready to act if the signal for global warming becomes clearer. To prepare, national governments should launch major programs of conservation to save native gene pools in seed banks, clone banks, or ex situ plantations."*

The signal for global warming is now clear, but in the United States, we have not prepared. The reason for inaction is that many politicians ask whether forest species like the Mexican spruces and *Picea breweriana* have any real value? Why should we save them? The ancient Hindus and Buddhists had an answer, perhaps right, perhaps exaggerated. This story from India, for example, about a great scholar and physician, Jivak, inscribed on the wall of the National Botanic Institute, Lucknow, India.

> *"Jivak, the ancient physician, when asked by his teacher to bring a plant that was quite useless, returned empty-handed, remarking that there was no such plant."*

However, the value of a species – the value of a forest – need not be solely economic – a piece of wood or a log that we can sell, a medicine that we can market.

The greatest value is ecologic. Forests provide indispensable ecosystem services. They sequester carbon from the atmosphere, and that kept the carbon dioxide level in balance at roughly 280 ppm for millions of years. It is common knowledge that forests can offset the emissions from burning fossil fuels, but in the short term, declining forests may contribute to the problem of global warming. Today, much of the carbon added to the atmosphere comes from forests (Bowman et al. 2009). Burning forests to clear land for pastures, or wildfires, accounts for a significant share of the carbon added to the atmosphere (Zarin 2012), at least 10% to 20% of global emissions. And as forests decline directly or indirectly because of global warming, they will add more carbon to the atmosphere and sequester less; in the forests of northern California, carbon emissions from fire are predicted to more than double by the year 2085 (Westerling et al. 2011). The climate projections presented in *Figures 5* and *7* carry an ominous message that is not just about the North American spruces. The Mexican spruces are not the only species that will be endangered. México alone has, perhaps, more than 3000 woody species, and about 40% of these may be endemic (Rzedowski 1986). And beyond Mexico, species throughout the world are at risk. Assisted colonization is needed not only for rare species, but also for common species. Furthermore, assisted colonization is needed not just to prevent extinction of species and loss of genetic diversity; more importantly, it will be needed to *maintain forest cover* – call it restoration ecology.

The world may be embarking on a positive feedback cycle – a dangerous runaway situation (*Figure 8*). The increase in carbon dioxide from burning fossil fuels has resulted in global warming, which will lead to forest decline. Forest decline has two effects: less carbon will be sequestered and the fuel load will increase. Higher fuel load and higher temperatures lead to more and hotter fires, and that leads to still higher levels of carbon dioxide, accelerating global warming.

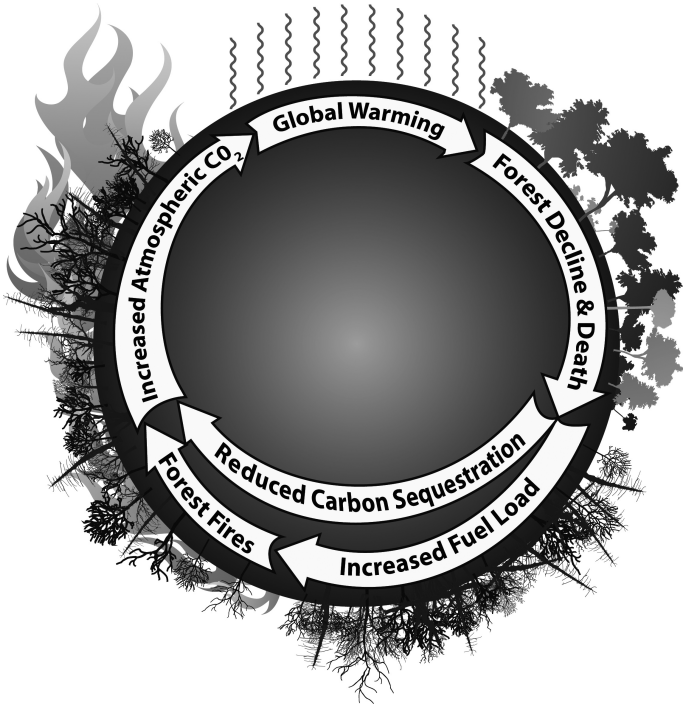



*Figure 8. Hypothesized positive feedback loop leading to increased $CO_2$ through forest decline and wildfire*

Global climate change confronts the world with a massive environmental crisis, and conserving some semblance of forest cover is a critical problem for the survival of civilization; for our survival, not just survival of the spruces. Although I realize this is alarmist, I am sincerely frightened for the future of our world. It may not be an exaggeration to liken the effects of global warming to the Four Horsemen of the Apocalypse. Coastal areas will flood, displacing millions of people. Agricultural productivity will decline in many areas, resulting in local famine. With the refugees displaced by flood and famine will come pestilence, and many countries will believe that war is their only option to gain needed resources – and they will have nothing to lose.

Jose Ortega y Gassett (1914) said it poetically in an essay he wrote about seeing the forest through the trees:

*"I am myself and what is around me, and if I do not save it, it shall not save me."*

Foresters have skills in planting and managing forests, and a vital role to play in stabilizing our environment. This is a century when foresters, in their role as restoration ecologists, will be urgently needed.

**Acknowledgments:** None of the work on the North American spruces reported here would have happened without the Forest Genetic Resources Working Group/North American Forest Commission/Food and Agricultural Organization of the United Nations. The research was carried out in cooperation with many individuals mentioned in the text, especially Celestino Flores-López, Gerald E. Rehfeldt, Cuauhtémoc Sáenz-Romero, Teobaldo Eguiluz-Piedra, Manuel Mápula-Larreta, Barry Jaquish, and my late colleague Basilio Bermejo-Velázquez.
The contents of the present paper were presented in an invited lecture at the University of West Hungary in September 2011, supported by the European Union and the European Social Fund under the grant agreement no. TÁMOP-4.2.2.-08/1-2008-0020.

## REFERENCES

AXELROD, D.I. (1987): The Late Oligocene Creede flora, Colorado. University of California Publications in Geological Sciences 130: 1–235.

BOWMAN, D. M. – BALCH, J. K. – ARTAXO, P. – BOND, W. J. – CARLSON, J. M. – COCHRANE, M.A. – D'ANTONIO, C.M. – DEFRIES, R.S. – DOYLE, J.C. – HARRISON, S.P. – JOHNSTON, F.H. – KEELEY, J.E. – KRAWCHUK, M.A. – KULL, C.A. – MARSTON, J.B. – MORITZ, M.A. – PRENTICE, I.C. – ROOS, C.I. – SCOTT, A.C. – SWETNAM, T.W. – VAN DER WERF, G.R. – PYNE, S.J. (2009): Fire in the Earth system. Science 324: 481–484.

BURNS, R.M. – HONKALA, B.H. (1990): Silvics of North America. Volume 1, Conifers. Agriculture Handbook 654. USDA, Forest Service, Washington, DC, USA.

CLISBY, K.H. – SEARS, P.B. (1955): Palynology in southern North America. Part III: microfossil profiles under Mexico City correlated with the sedimentary profiles. Bulletin of the Geological Society of America 66: 511–520.

FARJON, A. (2001): World checklist and bibliography of conifers, second edition. Kew, United Kingdom: Royal Botanic Gardens.

GILPIN, M.E. – SOULÉ, M.E. (1986): Minimum viable populations: processes of species extinction. In: SOULÉ, M. E (ed.): Conservation biology: the science of scarcity and diversity. Sinauer Associates, Sunderland, Massachusetts, USA, p. 19–34.

JARAMILLO–CORREA, J.P. – BEAULIEU, J. – LEDIG, F. T. – BOUSQUET, J. (2006): Decoupled mitochondrial and chloroplast DNA population structure reveals Holocene collapse and population isolation in a threatened Mexican-endemic conifer. Molecular Ecology 15: 2787–2800.

LEDIG, F.T. – JACOB-CERVANTES, V. – HODGSKISS, P.D. – EGUILUZ–PIEDRA, T. (1997): Evolution and divergence among populations of a rare Mexican endemic, Chihuahua spruce, following Holocene warming. Evolution 51: 1815–1827.

LEDIG, F.T. – BERMEJO-VELÁZQUEZ, B. – HODGSKISS, P. D. – JOHNSON, D. R. – FLORES, C. – JACOB-CERVANTES, V. (2000): Mating system and genic diversity in Martínez spruce, an extremely rare endemic of México's Sierra Madre Oriental: an example of facultative selfing and survival in interglacial refugia. Canadian Journal of Forest Research 30: 1156–1164.

LEDIG, F.T. – MÁPULA–LARRETA, M. – BERMEJO–VELÁZQUEZ, B. – FLORES–LÓPEZ, C. – REYES–HERNÁNDEZ, V. – CAPÓ–ARTEAGA, M. A. (2000): Locations of endangered spruce populations in México and the demography of *Picea chihuahuana*. Madroño 47: 71–88.

LEDIG, F.T. – HODGSKISS, P.D. – JACOB–CERVANTES, V. (2002): Genic diversity, the mating system, and conservation of a Mexican subalpine relict, *Picea mexicana* Martínez. Conservation Genetics 3: 113–122.

LEDIG, F.T. – HODGSKISS, P.D. – KRUTOVSKII, K. V. – NEALE, D.B. – EGUILUZ–PIEDRA, T. (2004): Relationships among the spruces (*Picea*, Pinaceae) of southwestern North America. Systematic Botany 29: 275–295.

LEDIG, F.T. – HODGSKISS, P.D. – JOHNSON, D.R. (2005): Genic diversity, genetic structure, and the mating system of Brewer spruce (Pinaceae), a relict of the Arcto-Tertiary forest. American Journal of Botany 92: 1975–1986.

LEDIG, F.T. – KITZMILLER, J. H. (1992): Genetic strategies for reforestation in the face of global climate change. Forest Ecology and Management 50:153–169.

LEDIG, F.T. – REHFELDT, G. E. – SÁENZ–ROMERO, C. – FLORES–LÓPEZ C. (2010): Projections of suitable habitat for rare species under global warming scenarios. American Journal of Botany 97: 970–987.

LEDIG, F.T. – REHFELDT, G.E. – JAQUISH, B. (2012): Projections of suitable habitat under climate change scenarios: implications for trans-boundary assisted colonization. American Journal of Botany 99: 1217–1230.

LOZANO–GARCÍA, M.S. – ORTEGA–GUERRERO, B. – CABALLERO–MIRANDA, M. – URRUTIA-FUCUGAUCHI, J. (1993): Late Pleistocene and Holocene paleoenvironments of Chalco Lake, central Mexico. Quaternary Research 40: 332–342.

MÁTYÁS, Cs. (2006): Migratory, genetic and phenetic response potential of forest tree populations facing climate change. Acta Silv. Lign. Hung. 2: 39–47

MÜLLER–USING, B. – ALANÍS–FLORES, G. (1984): Nuevos registros del pinabete de Chihuahua (*Picea chihuahuana* Martínez) en Nuevo León propuesta para la protección legas de dos áreas de especial interés ecológico. In Reunión Regional de Ecología Norte, 25, 26 y 27 de abril 1984, Monterrey, Nuevo León, México. Secretaría de Desarrollo Urbano y Ecología, Subecretaría de Ecología, México, p. 130-132. [New records of Chihuahua spruce (*Picea chihuahuana* Martínez) in Nuevo León proposed for legal protection in two areas of special ecological interest.] (in Spanish)

MÜLLER–USING, B. – LÄSSIG, R. (1986): Zur Verbreitung der Chihuahua-Fichte (*Picea chihuahuana* Martinez) in Mexiko. Mitteilungen der Deutschen Dendrologischen Gesellschaft 76: 157–169.

NEI, M. – LI, W. H. (1985): Mathematical model for studying genetic variation in terms of restriction endonucleases. Proceedings of the National Academy of Sciences USA 76: 5269–5273.

ORTEGA Y GASSET, J. (1914): Meditaciones del Quijote. Publicaciones Residencia de Estudios, series II, Vol. 1, Madrid.

PIMM, S.L. – RUSSELL, G. J. – GITTLEMAN, J.L. – BROOKS, T.M. (1995): The future of biodiversity. Science 269: 347–350.

SACCHERI, I. – KUUSSAARI, M. – KANKARE, M. – VIKMAN, P. – FORTELIUS, W. – HANSKI, I. (1998): Inbreeding and extinction in a butterfly metapopulation. Nature 392: 491–494.

SÁENZ–ROMERO C. – REHFELDT, G.E. – CROOKSTON, N.L. – DUVAL, P. – ST–AMANT, R. – BEAULIEU, J. – RICHARDSON, B. A. (2010): Spline models of contemporary, 2030, 2060 and 2090 climates for Mexico and their use in understanding climate-change impacts on the vegetation. Climatic Change 102: 595–623.

SOLOMON, S. – QIN, D. – MANNING, M. – CHEN, Z. – MARQUIS, M. – AVERYT, K.B. – TIGNOR, M. – MILLER, H.L. (2007): Climate change 2007: the physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, New York, USA.

WARD, P.D. (1995): The end of evolution: a journey in search of clues to the third mass extinction facing planet earth. Bantam Books, New York, USA.

WESTEMEIER, R.L. – BRAWN, J.D. – SIMPSON, S.A. – ESKER, T.L. – JANSEN, R.W. – WALK, J.W. – KERSHNER, E.L. – BOUZAT, J.L. – PAIGE, K.N. (1998): Tracking the long-term decline and recovery of an isolated population. Science 282: 1695–1698.

WESTERLING, A.L. – BRYANT, B.P. – PREISLER, H.K. – HOLMES, T.P. – HIDALGO, H.G. – DAS, T. – SHRESTHA, S.R. (2011): Climate change and growth scenarios for California wildfire. Climatic Change 109 (Supplement 1): S445–S463.

WHEELER, N.C. – HEHNEN, M. T. (1993): Taxol: a study in technology commercialization. Journal of Forestry 91(10): 15–18.

WILLIAMS, J.W. – JACKSON, S.T. (2007): Novel climates, no-analog communities, and ecological surprises. Frontiers in Ecology and the Environment 5: 475–482.

WOLFE, J. A. (1964): Miocene floras from Fingerrock Wash southwestern Nevada. United States Geological Survey Professional Paper no. 454-N.

ZARIN, D. J. (2012): Carbon from tropical deforestation. Science 336: 1518–1519.

VERSITA DOI: 10.2478/v10303-012-0006-3 Acta Silv. Lign. Hung., Vol. 8 (2012) 75–86

# Genetic Variability of two *Fagus sylvatica* (L.) Populations in the South-Western Edge of the Panonnian Plain[1]

Gregor BOŽIČ* – Lado KUTNAR

Slovenian Forestry Institute, Ljubljana, Slovenia

**Abstract** – Two beech populations at altitudes of 273 m (Vrhovo) and 657 m (Kozarje), growing on significantly different forest site, in different ecological conditions and with different forest-management history were selected on Gorjanci Mountains and genetically investigated by means of isoenzyme analysis. A cline from minor to major allelic polymorphism was revealed at gene loci Aco-B, Idh-A and 6-Pgdh-A. At locus Skdh-A, allele $A_4$ was observed only at the lower altitude population Vrhovo as a heterozygote genotype Skdh-$A_{34}$ (12%). The Kozarje population of beech had a slightly higher genetic multiplicity, a slightly higher effective number of alleles per locus, higher observed heterozygosity and a higher level of intrapopulation differentiation. Statistically significant differences in the occurrence of alleles between populations were found at 5 of 16 loci. The average genetic distance of Gregorius (1974) was 6.1%. European beech populations analyzed from different altitudes on Gorjanci Mountains appeared to be genetically differentiated. The results presented are also discussed in the light of the climate change impact on forests.

**European beech / isoenzyme / genetic polymorphism / genetic differentiation / forest site / Slovenia**

**Kivonat – Két bükk populáció genetikai változatossága a Kárpát medence délnyugati peremén.** Két jelentősen eltérő termőhelyen tenyésző bükk populációt izoenzim analízis segítségével genetikailag elemeztünk (Vrhovo, 273 m és Kozarje 657 m). A Gorjáná hegyekben tenyésző populációk ökológiai feltételei és erdőművelési múltja különbözők. Az Aco-B, Idh-A and 6-Pgdh-A lokuszokon az allél polimorfizmus klínje olt megfigyelhető. Skdh-A lokuszon az $A_4$ allél csak az alacsonyabb fekvésű populációban (Vrhovo) volt kimutatható, mint Skdh-$A_{34}$ heterozigóta genotípus. A Kozarje-i bükkös genetikai változatossága kissé nagyobb volt, magasabb effektív lokuszonkénti allélszámmal, magasabb heterozigozissal és magasabb populáción belüli differenciálódással. Az allélek összetétele a 16 génhely közül 5 esetében szignifikáns eltérést mutatott. Az átlagos Gregorius-féle genetikai távolság 6.1% volt. A különböző tengerszint feletti magasságban tenyésző populációk genetikailag eltérőnek bizonyultak. a tanulmány az eredményeket a klímaváltozással összefüggésben is elemzi.

**közönséges bükk / izoenzimek / genetikai polimorfizmus / genetikai differenciáltság / Termőhelyi különbségek / Szlovénia**

[1] Paper presented at the IUFRO Fagus meeting 2010, and also available at *Šumarski list*, Zagreb

* Corresponding author: gregor.bozic@gozdis.si; Večna pot 2, SI-1000 LJUBLJANA

## 1 INTRODUCTION

Slovenia is one of the most forested countries in Europe, with a total coverage of 1.184.369 ha or 58.45 of total area (SFS 2012). Forests are well preserved, and sustainable, semi-natural and multifunctional management has been traditionally and legally incorporated into the forestry practice. After the primary succession in the postglacial period, the larger part of the Slovenian territory was overgrown by forests, above all by beech and fir-beech forests (Šercelj 1996). According to Perko (2007), 70% of forests in Slovenia grow on potential beech (44%), fir-beech (15%) or beech-oak (11%) sites. European beech is autochthonous, dominant and the most economically and ecologically important tree species in Slovenia. It occurs on all terrain positions and slope orientations on calcareous, silicate and mixed calcareous-silicate bedrocks (Urbančič et al. 2005), from hills (150 m a.s.l.) to the subalpine belt (1650 m a.s.l.) (Dakskobler 2008).

In the 1990s, the genetic variation of beech populations in Slovenia and, in comparison, with its variation in Central and Southeast Europe were studied using isoenzyme gene markers (Paule et al. 1995, Brus 1999, Brus et al. 1999, Gömöry et al. 1999). Results of these studies have shown the existence of genetic differences between beech populations from the north-western part and those from the eastern part of Balkan Peninsula. The obtained results further supported the hypothesis that during the ice ages European beech was present in microrefugia at the south-eastern periphery of Alps and on the territory of present-day Slovenia (Brus et al. 2000, Brus 2010). Findings were confirmed by the study of Magri et al. (2006), which analysed large palaeobotanical and genetical data of common beech in Europe. The territory of present-day Slovenia was one of the main source areas for the post-glacial development of beech and supposedly the most important glacial refugia for its re-colonisation in Europe. The development of beech forests allowed a possibility that the European beech in the territory of present-day in Slovenia went through a genotypic specialisation that also resulted in locally adapted races or ecotypes.

As genetic diversity is a fundamental and critical part of biodiversity the results of the genetic variation constituted a professional basis for the conservation of natural genetic variability *in-situ*. The objective is to maintain continuous evolution of a tree population. Natural regeneration of adult beech stands in Southeast Europe enables a sustainable conservation of gene resources. The decision for the choice of gene resources and form of their protection should be based on all available information, including that which can be obtained from genetic inventories on isoenzymes. In general it has been found that it is more worth to designate for conservation relatively large units than many smaller units within gene conservation practices (Paule 1995). Such gene reserve forests should cover at least 100 ha in order to contain sufficient genetic variability. But for small, locally adapted populations it may be better to establish a large number of smaller reserves (von Wühlisch 2008). However, in its natural distribution area, very little is known about the patterns of population genetic variation in geographically smaller but heterogeneous areas.

Genetic study by Comps et al. (1991) based on beech populations from Croatia originated from different climatic regions, soils and forest communities has shown differences between highland and lowland populations within the Mediterranean region as well as between populations belonging to the forest community *Seslerio-Fagetum* on carbonate soils and other forest communities. The results from beech provenance trails represent great phenotypic and phenological differences among Slovenian provenances from various elevations, expositions, climatic conditions (Brinar 1971, Sittler 1981, Robson et al. 2011).

In order to look for a possible pattern of beech population genetic variation in geographically smaller and heterogeneous areas, two autochthonous *Fagus sylvatica* L. populations were selected on Gorjanci Mountains (southeast Slovenia) to avoid any difference in population history and genetically analysed by isoenzymes gene markers.

## 2 STUDY AREA

Gorjanci Mountains are 45–50 km long and 18–22 km wide mountain massive at the south-western edge of the Panonnian plains. The massive is characterized by a dynamic relief under influence of two major European geographical and climate units: the Panonnian basin on the north-east and the Dinaric Mountains on the west to south-west (Kutnar et al. 2002). At the bottom of Gorjanci Mountains, beech (*Fagus sylvatica* L.) forests with mixtures of different tree species, including *Quercus petraea* (Matt.) Liebl., *Carpinus betulus* L., *Picea abies* (L.) H. Karst., *Acer pseudoplatanus* L., *Acer campestre* L. and others, cover the major part of the forested area in the transition from the submontane belt to the lowlands. Due to their vicinity to human settlements, they have always sustained heavy anthropogenic impacts. In particular, there are many coppiced forests close to farms. Some of these sites were converted to coniferous monocultures, and many of them were even transformed into agricultural use. In contrast, the human impact is not so pronounced at higher zone of Gorjanci Mountains, and more or less pure beech stands extend over a larger area. In this area, the share of the coppice and spruce monoculture forests are very low, and forest land-use is prevailing (Marinček – Čarni 2002).

Research plots were selected at different autochthonous beech sites of Gorjanci Mountains, both belonging to Natura 2000 habitat type of 91K0 Illyrian *Fagus sylvatica* forests (Kutnar et al. 2011) in the hilly and the mountainous vegetation zone. Two populations of beech at Vrhovo and Kozarje were sampled in "selected" category of forest reproductive material in beech seed stands of provenance Ustraški boršt/Cerov Log at altitude of 273 m (ident. number GSO 5.0222) and provenance of Gorjanci/Kozarje at altitude of 657 m (ident. number GSO 5.0216), respectively (Kraigher et al. 2012). The Vrhovo population belongs to forest of beech and sessile oak with ivy (*Hedero-Fagetum* var. geogr. *Epimedium alpinum,* syn. *Querco petraeae-Fagetum*). The Kozarje population belongs to the Praedinaric mountain beech forest with dead nettles (*Lamio orvalae-Fagetum* var. geogr. *Dentaria polyphyllos*).

*Table 1. Survey of site characteristics of sampled beech populations on Gorjanci Mountains*

| Population | Altitude | Latitude E | Longitude N | Average annual precipitation * | Average annual temperature * | Dominant soil types |
|---|---|---|---|---|---|---|
| | (m) | | | (mm) | (°C) | |
| Vrhovo | 273 | 45°48'25" | 15°18'11" | 1300–1400 | 10–12 | Distric Cambisol |
| Kozarje | 657 | 45°48'22" | 15°27'33" | 1400–1500 | 8–10 | Eutric Cambisol, Rendzic Leposol |

* Reference period: 1971–2000 (ARSO 2010)

The research site at Vrhovo is overgrown by *Fagus sylvatica* L. (84%), *Quercus petraea* (Matt.) Liebl. (9%), *Picea abies* (L.) H. Karst. (3%), *Acer pseudoplatanus* L. (3%) and *Carpinus betulus* L. (1%) with an estimated stand growing stock of 205 $m^3$ ha and incremenent of 5.6 $m^3$ ha yr, and at Kozarje by *Fagus sylvatica* L. (90%), *Acer pseudoplatanus* L. (4%), *Prunus avium* L. (4%), *Carpinus betulus* L. (1%) and other broadleaves (1%) with a stand growing stock of 310 $m^3$ ha and incremenent of 9,7 $m^3$ ha yr (Simončič et al. 2007). The distance between sampled beech populations was 13 km with 384 m in altitude.

## 3 MATERIAL AND METHODS

### 3.1 Sampling

At the research sites Vrhovo and Kozarje on Gorjanci Mountains, randomly were sampled 100 beech trees for genetic testing over an area of 3.5 ha. In the winter period of 2005/2006, we took a branch with dormant buds from each of the sampled trees, which were used for the extraction of enzymes. The buds were preserved until analysis in test tubes at a temperature of -20 °C. The sample size was 50 adult trees at each site.

### 3.2 Analysis of isoenzymes

The genetic variability of the two sampled beech populations was analyzed by means of isoenzyme gene markers using starch electrophoresis as the separation method. Enzyme extraction from dormant buds, electrophoresis conditions and staining, and enzyme visualization was performed according to Konnert et al. (2004). Ten isozyme systems coded by sixteen gene loci were surveyed (Aat syn. Got, Aco, Idh, Mdh, Mnr, Per, Pgi, Pgm, Skdh, 6-Pgdh). The genetic interpretation of banding patterns followed Müller-Starck et al. (2001). The laboratory analyses were performed in the framework of the research tasks of the project Carbon dynamics in natural beech forests (L4-6232) in February 2006. The laboratory work was done at the Bayerische Amt für forstliche Saat - und Pflanzenzucht in Teisendorf, Germany.

The results of the isoenzyme analyses were evaluated by the relative allele frequencies, calculated on different gene loci after diploid tree genotypes. Allelic structures in each gene loci were estimated by four allelic profiles according to Finkeldey (1993). Genetic multiplicity was measured by the highest possible number of different alleles ($M_{max}$) and the average number of alleles per polymorphic locus (A/L). Genetic diversity was quantified using parameters of the observed level of heterozygosity ($H_o$) and the conditional heterozygosity ($H_c$; Gregorius et al. 1986), effective number of alleles per locus ($\nu$; Gregorius 1978, 1987), hypothetical gametic multilocus diversity ($\nu_{gam}$) and the level of genetic differentiation among individuals within a population ($\delta_T$; Gregorius 1987) which, with larger samples, is the same as the share of expected heterozygosity, created by random mating or panmixia ($H_e$; Nei 1973). For each polymorphic locus, $\chi^2$ tests of deviation from the corresponding Hardy-Weinberg expected genotypic structures as well as Hardy-Weinberg heterozygosity at the level $\alpha = 0.05$ were carried out to test whether the observed genotypic structure deviates from panmixia and whether there was a significant excess or deficiency of heterozygotes in a population. The degree of differentiation between populations was measured with $\chi^2$ tests of homogeneity among allele frequencies for particular gene loci at the level $\alpha = 0.05$ and genetic distances ($d_0$) proposed by Gregorius (1974). All computation was performed with GSED software (Gillet 1998) for analyzing genetic structures from electrophoresis data.

## 4 RESULTS

The results of genetic comparison of sampled beech populations at Vrhovo and Kozarje are shown in Tables 2 and 3 for 16 isozyme gene loci. Minor polymorphism in both populations was observed at eight loci: Aat-A, Aco-A, Mdh-A, Mnr-A, Pgi-B, Skdh-A, 6-Pgdh-B, 6-Pgdh-C with the frequency of the major allele always higher than 80%. Four loci (Aat-B, Mdh-C, Per-B, Pgm-A) showed a clear major polymorphism with the same predominant allele in both populations. On locus Mdh-B the frequencies of the major allele were from 75% to 77%,

with frequencies of the second predominant allele from 10% to 17%. A distinct transition between low and high levels of polymorphism in the test populations were found at three loci. In the Vrhovo population (from a lower altitude), minor polymorphism was expressed at loci Aco-B and Idh-A and major polymorphism at locus 6-Pgdh-A. In the Kozarje population (from a higher altitude), a clear minor polymorphism was expressed at locus 6-Pgdh-A and clear major polymorphism at loci Aco-B and Idh-A. For example: alleles Aco-$B_2$, Idh-$A_2$ and 6-Pgdh-$A_4$ in the Vrhovo population reached 8%, 12%, 28%, respectively, in the Kozarje population 20%, 25% and 11%, respectively. A clear differentiation between the two populations was also observed at locus 6-Pgdh-B, where the population from higher altitude of the mountain ridge revealed much higher frequency of the second dominant allele $B_1$ (16%) in comparison to the 6% frequency at the lower altitude population.

*Table 2. Relative allele frequencies in sampled beech populations Vrhovo and Kozarje on Gorjanci Mountains presented per gene loci*

| Locus | Allele | Vrhovo N = 50 | Kozarje N= 50 | Locus | Allele | Vrhovo N = 50 | Kozarje N = 50 |
|---|---|---|---|---|---|---|---|
| Aat-A | 1 | 0.030 | 0.050 | Mnr-A | 2 | 0.010 | 0.010 |
| | 2 | 0.970 | 0.950 | | 3 | 0.930 | 0.900 |
| Aat-B | 2 | 0.350 | 0.320 | | 4 | – | 0.030 |
| | 3 | 0.650 | 0.680 | | 5 | 0.060 | 0.060 |
| Aco-A | 2 | 0.970 | 0.990 | Per-B | 1 | 0.260 | 0.260 |
| | 3 | 0.030 | 0.010 | | 2 | 0.670 | 0.650 |
| Aco-B | 1 | – | 0.010 | | 3 | 0.070 | 0.090 |
| | 2 | 0.080 | 0.200 | Pgi-B | 2 | 0.990 | 0.990 |
| | 3 | 0.900 | 0.790 | | 3 | 0.010 | 0.010 |
| | 4 | 0.020 | – | Pgm-A | 2 | 0.200 | 0.240 |
| Idh-A | 2 | 0.120 | 0.250 | | 3 | 0.800 | 0.760 |
| | 3 | 0.880 | 0.750 | Skdh-A | 3 | 0.930 | 0.960 |
| Mdh-A | 1 | 0.020 | 0.040 | | 4 | 0.060 | – |
| | 3 | 0.980 | 0.960 | | 5 | 0.010 | 0.040 |
| Mdh-B | 1 | 0.080 | 0.060 | 6-Pgdh-A | 2 | 0.720 | 0.880 |
| | 2 | 0.050 | 0.020 | | 3 | – | 0.010 |
| | 3 | 0.770 | 0.750 | | 4 | 0.280 | 0.110 |
| | 4 | 0.100 | 0.170 | 6-Pgdh-B | 1 | 0.060 | 0.160 |
| Mdh-C | 1 | 0.310 | 0.270 | | 2 | 0.940 | 0.840 |
| | 2 | 0.690 | 0.730 | 6-Pgdh-C | 1 | 0.820 | 0.790 |
| | | | | | 3 | 0.020 | 0.010 |
| | | | | | 4 | 0.100 | 0.190 |
| | | | | | 5 | 0.060 | 0.010 |

As enzyme genotypes represent the enzymes that catalyse the metabolism of plants, the genotypes of trees in sampled populations were also studied. Homozygous and heterozygous types that are much more frequent in the Vrhovo sample than in the Kozarje sample are: Idh-$A_{33}$ (80% vs. 56%), Aco-$B_{33}$ (80% vs. 60%), 6-Pgdh-$B_{22}$ (90% vs. 72%), Pghd-$A_{24}$ (44% vs. 18%), Pgdh-$C_{14}$ (34% vs. 20%) and Mdh-$C_{12}$ (50% vs. 38%). Heterozygous or homozygous types much more frequent in the Kozarje sample than in the Vrhovo sample are: Idh-$A_{23}$ (38% vs. 16%), Aco-$A_{23}$ (36% vs. 16%), 6-Pgdh-$B_{12}$ (24% vs. 8%), Mdh-$B_{34}$ (26% vs. 12%), 6-Pgdh-$C_{15}$ (12% vs. 2%) and 6-Pghd-$A_{22}$ (78% vs. 50%). Remarkable differences in allelic and genotypic frequencies were observed at gene locus Skdh-A. The allele $A_4$ and

heterozygous genotype $A_{34}$ are stand-dependent. It was found at frequencies of 6% and 125 in the lower elevated Vrhovo population, respectively, but not in the higher elevated Kozarje population.

On the study area, we observed 43 different alleles ($M_{max}$) which correspond to 2.69 alleles per polymorphic locus in both sampled populations (Table 2). The mean numbers of alleles per locus (A/L) were 2.50 at Vrhovo and 2.56 at Kozarje population (Table 3), which is equivalent to a ratio of 1:1.02 The mean values of the effective number of alleles per locus (ν) range from 1.31 (Vrhovo) to 1.34 (Kozarje) and correspond to a ratio of 1:1.05. The numbers of genetic variants of the 16 loci gametic types that the group of 50 sample trees in the populations ($\nu_{gam}$) could have produced ranges from 104 (Vrhovo) to 157 (Kozarje) and corresponds to a ratio of 1:1.51. Although the calculated values ($\nu_{gam}$) have only an indicative character, this comparison suggests greater potential of the group of beech trees at the Kozarje site than that of the group of beech trees at the Vrhovo site for the production of genetically different gametes, which will be subject to genetic variation in new generations.

*Table 3. Genetic variability parameters for 16 polymorphic loci in the sampled beech populations, Vrhovo "Vrh" and Kozarje "Koz" on Gorjanci Mountains. Average number of alleles per locus (A/L), allelic diversity (ν), intrapopulational differentiation ($\delta_T$), observed and conditional heterozygosity ($H_o$, $H_c$), allelic genetic distance ($d_0$) and value of $\chi^2$ test of the homogeneity of genetic structures by loci and significance level $\alpha = 0.05$ (*), $\alpha = 0.01$ (**)*

| Locus | A/L | | ν | | $\delta_T$ (%) | | $H_o$ (%) | | $H_c$ (%) | | $d_0$ (%) | $\chi^2$ value |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Vrh | Koz | Vrh | Koz | Vrh | Koz | Vrh | Koz | Vrh | Koz | | |
| Aat-A | 2 | 2 | 1.06 | 1.11 | 5.9 | 9.6 | 6.0 | 6.0 | 100 | 60.0 | 2.0 | n.s. |
| Aat-B | 2 | 2 | 1.84 | 1.77 | 46.0 | 44.0 | 42.0 | 44.0 | 60.0 | 68.8 | 3.0 | n.s. |
| Aco-A | 2 | 2 | 1.06 | 1.02 | 5.9 | 2.0 | 6.0 | 2.0 | 100 | 100 | 2.0 | n.s. |
| Aco-B | 3 | 3 | 1.22 | 1.51 | 18.5 | 33.9 | 20.0 | 38.0 | 100 | 90.5 | 13.0 | 8.859 * |
| Idh-A | 2 | 2 | 1.27 | 1.60 | 21.3 | 37.9 | 16.0 | 38.0 | 66.7 | 76.0 | 13.0 | 4.775 * |
| Mdh-A | 2 | 2 | 1.04 | 1.08 | 4.0 | 7.8 | 4.0 | 8.0 | 100 | 100 | 2.0 | n.s. |
| Mdh-B | 4 | 4 | 1.64 | 1.68 | 39.2 | 40.9 | 34.0 | 42.0 | 73.9 | 84.0 | 7.0 | n.s. |
| Mdh-C | 2 | 2 | 1.75 | 1.65 | 43.2 | 39.8 | 50.0 | 38.0 | 80.6 | 70.4 | 4.0 | n.s. |
| Mnr-A | 3 | 4 | 1.15 | 1.23 | 13.3 | 18.7 | 14.0 | 16.0 | 100 | 80.0 | 3.0 | n.s. |
| Per-B | 3 | 3 | 1.92 | 2.01 | 48.3 | 50.7 | 46.0 | 48.0 | 69.7 | 68.6 | 2.0 | n.s. |
| Pgi-B | 2 | 2 | 1.02 | 1.02 | 2.0 | 2.0 | 2.0 | 2.0 | 100 | 100 | 0.0 | n.s. |
| Pgm-A | 2 | 2 | 1.47 | 1.57 | 32.3 | 36.8 | 32.0 | 32.0 | 80.0 | 66.7 | 4.0 | n.s. |
| Skdh-A | 3 | 2 | 1.15 | 1.08 | 13.3 | 7.8 | 14.0 | 8.0 | 100 | 100 | 6.0 | 7.848 * |
| 6-Pgdh-A | 2 | 3 | 1.68 | 1.27 | 40.7 | 21.6 | 44.0 | 20.0 | 78.6 | 83.3 | 17.0 | 10.010 ** |
| 6-Pgdh-B | 2 | 2 | 1.13 | 1.37 | 11.4 | 27.2 | 8.0 | 24.0 | 66.7 | 75.0 | 10.0 | 4.137 * |
| 6-Pgdh-C | 4 | 4 | 1.51 | 1.46 | 34.3 | 31.7 | 38.0 | 36.0 | 90.5 | 100 | 9.0 | n.s. |
| Gene pool | 2.50 | 2.56 | 1.31 | 1.34 | 23.7 | 25.8 | 23.5 | 25.1 | 85.5 | 82.7 | 6.1 | |

The average values of observed heterozygosities ($H_o$) for the 16 loci gene pool varied between 23.55 in Vrhovo sampled population to 25.1% in the Kozarje sampled population, which corresponds to a ratio of 1:1.06. As shown in Table 3, the $H_o$ values are not uniform. The level of observed heterozygosity explicitly differs between the test populations from

Vrhovo and Kozarje at 5 loci namely at Aco-B (20% vs. 38%), Idh-A (16% vs. 38%), Mdh-C (50% vs. 38%), 6-Pgdh-A (44% vs. 20%) and 6-Pgdh-B (8% vs. 24%). The mean of $H_c$ value for two populations is 84.1%. As $H_c$ values are independent of the underlying allele frequencies, the calculated mean indicated to a slightly lack of heterozygotes as compared to the maximum attainable value which is equal to 100%. The average levels of genetic (allele) differentiation $\delta_T$ in the sampled populations of beech range from 23.7% (Vrhovo) to 25.8% (Kozarje), which corresponds to a ratio of 1:1.09 and indicates larger genetic differentiation among individuals at the higher elevated site than among individuals at the lower elevated site. No significant deviations of observed genotypes from expected genotypes were found at 15 of the total of 16 analysed gene loci when applying $\chi^2$ test ($\alpha = 0.05$). The only exception was detected at the locus 6-Pgdh-B at Vrhovo population where no statistically significant excess of homozygotes over Hardy-Weinberg expectations was confirmed. The observed frequencies of genotypes in both populations of beech agree with Hardy-Weinberg expectations under random mating at the level of risk $\alpha = 0.05$.

The degree of genetic differentiation between populations was analysed with $\chi^2$ tests of homogeneity among allele frequencies for particular gene loci at the level $\alpha = 0.05$ and $\alpha = 0.01$. The results have shown significant differences in allelic (genetic) structures between sampled populations of beech at 5 out of 16 analysed gene loci (Table 3). The sampled populations of beech did not share 17% of their alleles at locus 6-Pgdh-A and 13% of their alleles at loci Aco-B and Idh-A as well as 30% of their genotypes at locus 6-Pgdh-A, 24% of their genotypes at locus Idh-A and 12% of their genotypes at locus Skdh-A (genotypes values not introduced in Table 3). The mean value of Gregorius (1974) allelic and genotypic genetic distances ($d_0$) amounts to 6.1% and 11.4%, respectively, shows that the genetic differences of two local sampled beech populations from different sites on Gorjanci Mountains are comparatively high.

## 5 DISCUSSION AND CONCLUSIONS

Two very specific beech populations on Gorjanci Mountains, growing on significantly different forest sites, in different ecological conditions and with different forest-management history were selected. The Vrhovo population was sampled in the hilly area and belongs to site of *Hedero-Fagetum*. The Kozarje population was sampled in the mountainous zone and belongs to site *Lamio orvalae-Fagetum*. We have analyzed 50 trees per sample to capture the most of the genetic variation present at the different sites. If population history is corrected for (as we can guess it is here as they are from the same Mountain range), the difference in allele frequencies obtained in two sampled populations from different altitudes may better be explained by selection and adaptation rather than by population structure.

The observed genotypes of the two beech populations on Gorjanci Mountains were consistent with the expectations under random mating, and indicated that both sampled populations are randomly mating. At 16 polymorphic enzyme gene loci, 43 allelic variants were found in total, which correspond to 2.69 alleles per locus. Beech from the sampled local populations on Gorjanci Mountains seems to have a similar number of alleles per locus as the overall value for beech in Europe, according to Müller-Starck et al. (1992). This value also corresponds well with A/L = 2.8 obtained for Slovenia (Brus et al. 1999) and was higher than was reported for north-western Italy A/L = 2.12 (Belleti – Lanteri 1996) and Germany A/L = 2.51 (Konnert 1995). The average A/L values observed in the sampled beech populations on Gorjanci Mountains (2.50 and 2.56) show higher values than was determined for 13 beech populations in Slovenia (Brus et al. 1999) where A/L values varied in range from 2.08 to 2.42 and were close to the highest value obtained in the population of Luknja from Prečna close to

Novo mesto. Beech from Luknja originated from mainly southern-orientated steep limestone walls close to the Palaeolithic site from which beech charcoal from the late Pleistocene has been excavated (Osole 1990); it may represent a relict population that persisted in a sheltered micro-refugium where it was able to survive the last glaciations (Brus et al. 1999). Therefore it seems that no restriction in genetic multiplicity was determined in beech populations at Gorjanci Mountains.

The populations of European beech at Vrhovo and Kozarje which belong to sites of *Hedero-Fagetum* in the hilly area and *Lamio orvalae-Fagetum* in the mountainous zone, respectivetly are autochthonous. The higher altitude population at Kozarje showed higher genetic diversity than the lower altitude population at Vrhovo. Differences are evident in the frequencies of some alleles and genotypes, observed heterogosity and particularly in the hypothetical gametic multilocus diversities. The frequency of the allels 6-Pgdh-$B_1$, Aco-$B_2$ and Idh-$A_2$ at the higher elevated Kozarje population were more than twice as high as of the lower elevated stand. The frequency of the allele 6-Pgdh-$A_4$ was about three times higher in the (lower elevated) Vrhovo population than in the (higher elevated) Kozarje population. At locus Skdh-A, allele ($A_4$) was observed only in the Vrhovo population as a heterozygote genotype Skdh-$A_{34}$ (12%). The frequency of the genotype Skdh-$A_{34}$ significantly changed with altitude.

Our findings confirm findings from comparable studies of beech populations in Germany, related to climatic adaptation with an altitudinal transect from the base of the mountain to its top. In an inventory study on the isozyme genetic variation of beech stands growing at elevations between 350 m and 1250 m a.s.l. in the Black Forest (Mountain Schauinsland, southwest Germany), allele frequencies of different enzyme systems at gene loci Aco-B, Idh-A, Pgm-A were found to be changed (Löchelt – Franke 1995). Climate adaptation and manmade selection were considered to be causes. A detailed investigation of six autochthonous beech stands growing at elevations between 150 m and 660 m a.s.l. in the same mountain slope on Vogelsberg in central Germany showed that some genotypes are more frequent at the higher elevated stands than in lower elevated stands and vice versa (Sander et al. 2000).

Beech populations sampled from different sites on Gorjanci Mountains appeared to be genetically different. Genetic differentiation, as expressed by the proportion of alleles not shared between lower and higher elevated population was relatively high for the small geographical region studied ($d_0$ = 6.1%). For example, the genetic distances of beech populations in Germany (Bavaria) studied using the same analyzing method were reported to range between $d_0$ = 2.6% and $d_0$ = 10.9% (Konnert – Henkel 1997). The observed clines at gene loci Aco-B, Idh-A, 6-Pgdh-A and the allelic distances from 13% to 17% between sampled populations on Gorjanci Mountains could indicate that gene flow between sampled populations is limited or that specific site conditions in sampled locations demonstrate a possible effect of environmental adaptation to this gene loci.

Althought the obtained results do not allow us to infer any certain conclusions on possible adaptive role of the enzyme systems under analysis, also because of only two populations studied (one in each ecological conditions) and possible samplings mistakes on account of the large size of analysed samples, they verify the existence of genetic differentiation between two local beech populations on Gorjanci Mountains associated with variation of significantly different forest sites, different ecological conditions and different forest-management history. As the obtained results also confirm the similar patterns or correlations of the types mentioned in German studies focused in different ecological conditions in altitude gradient and in different parts of the beech distribution range (Löchelt – Franke 1995, Sander et al. 2000), the unknown selective pressure would have been considered as the causal agent for differentiation between European beech populations. However, to

prove the hypothesis of the interaction between disruptive natural selection and gene flow in development of among population genetic differentiation of beech further studies should be preformed.

The structure of the obtained genetic information may be the result of the sampled stands' high adaptations and specializations to specific homogenous environmental conditions in the investigated ecosystems and historical influences of anthropogenic impacts. Under constant environmental conditions, the higher genetic variability within populations means a higher fitness of the population. This situation remains optimal as long as the site conditions are not affected by climate changes and manmade selection to that extent that a process of adaptation to new site conditions is interrupted. Regarding the climate change impact on forests, the different studies predict the decrease of beech based on climate change scenarios in the future in Europe (Brzeziecki et al. 1995, Fotelli et al. 2002, Geßler et al. 2007, Mátyás et al. 2010, Kutnar – Kobler 2011).

For the Slovenian territory, the reduction of European beech sites was forecasted (Kutnar – Kobler 2011). In Central Europe (including Slovenia), beech forests are likely to be threatened because of beech's sensitivity to low water availability (Ellenberg 1996) and longer drought periods (Fotelli et al. 2002); and physiological performance, growth and the competitive ability of beech may be adversely affected by such changing climate conditions (Peuke et al. 2002, Geßler et al. 2007). Moreover, in Slovenia, the situation may be aggravated by the fact that the area of distribution of beech forests includes many sites with shallow limestone- and dolomite-derived soils with low water storage capacity. Beside different abiotic factors, the existence of beech is threatened by harmful pests and new tree diseases caused by different pathogen organisms (Ogris et al. 2008).

For Slovenia, the GIS models showed that under warmer conditions the shift of vegetation belts upwards could be expected (Kutnar – Kobler 2011). This means that *Fagus*-dominated communities in the submontane belt might eventually be replaced by oak-hornbeam communities, and the shift of tree-line to a higher elevation is predicted. As reported Mátyás et al. (2010) for the beech forests at the xeric limit in Southeast Europe (case Hungary), the very similar pattern might be expected at Gorjanci area where low-elevation beech forests (*Hedero-Fagetum* forests) might be threatened by the warming more than higher-elevation beech forests (*Lamio orvalae-Fagetum* forests). In more preserved, less degraded sites of *Hedero-Fagetum* forest, beech is well developed and has high competitive potential against other tree species. However, at more degraded sites of this forest community, where intensive human influence took place in the past, beech is not so competitive and its growth is less intensive, and the quality of beech stems is not so high. In such degraded sites of *Hedero-Fagetum*, some species of earlier forest development stages, like *Quercus petraea*, *Q. cerris* and *Carpinus betulus*, are more competitive and more frequent. In such human-degraded forms of this forest community, the coppice forest is more common. In such forest conditions, the lability of forest sites with the dominate Luvisol to moderately dystric Cambisol might have been aggravated by inappropriate forest management in the past, such as litter-raking, and intensive use without any management concept. Due to old settlements at the bottom of Gorjanci Mountains, the forests in the hilly zone have been under human impact. Thus, the relative high share of coppice, litter-raking forest sites planted by non-native tree species and other degraded forests in this area reflect the negative human impacts in the past. The mountain beech forests of *Lamio orvalae-Fagetum* at Gorjanci, compared to more degraded sites of *Hedero-Fagetum* forest, have a stable biocenotic structure. Even after larger clearings, all development stages of the forests can be renewed by beech trees. Due to their more remote and isolated location, the sites of these forests were not transformed into agricultural land.

As the ability of forest ecosystems is strongly related to the existence of intrapopulation genetic diversity, even in mixed natural stands, the stability is threatened as soon as the genetic diversity of one of the involved species is threatened; there is a great need for investigations of the genetic basis of forest ecosystems in geographically smaller but heterogeneous areas. Although the results of our study have to be confirmed by further genetic studies, attention to the transfer of beech reproductive material from higher to lower elevations of Gorjanci Mountains is recommended.

**Acknowledgments:** This study took place partially within the framework of research tasks of projects: V4-1140, V4-492, L4-6232 and research programme P4-0107 financed by the Slovenian Research Agency and co-financed by the Ministry for Agriculture and Environment. The authors are grateful to Dr. Monika Konnert (ASP, Teisendorf) for all her help in the genetics laboratory. Technical assistance of Sussane Nowak (ASP, Teisendorf) and Barbara Štupar (SFI, Ljubljana) is greatly appreciated. We also thank Mari Mette Tollefsrud, An Vanden Broeck and two anonymous reviewers for critical reading and valuable comments on the manuscript. The English language was checked by Terry Troy Jackson.

## REFERENCES

ARSO (2010): Environmental Atlas of Slovenia. Ministry of Agriculture and Environment RS, Slovenian Environment Agency. Ljubljana, Slovenia.
URL: *http://gis.arso.gov.si/atlasokolja/profile.aspx?id=Atlas_Okolja_AXL@Arso*

BELLETI, P. – LANTERI, S. (1996): Allozyme variation among European beech (*Fagus sylvatica* L.) stands in Piedmont, North-Western Italy. Silvae Genetica, 45: 33–37.

BRINAR, M. (1971): O ekološki in dedni pogojenosti razhajanja nekaterih morfoloških, fenoloških in anatomskih značilnosti naše bukve. [The ecological and genetic divergences of some morphological, phenological and anatomical characteristics of our beech]. Zbornik gozdarstva in lesarstva, 10: 5–64 (in Slovenian with German abstract)

BRUS, R. (1999): Genetska variabilnost bukve (*Fagus sylvatica* L.) v Sloveniji in primerjava z njeno variabilnostjo v srednji in jugovzhodni Evropi. [Genetic variation of the beech (*Fagus sylvatica* L.) in Slovenia and comparison with its variation in central and southeastern Europe]. Dissertation, Biotehniška fakulteta, Ljubljana, 130 p. (in Slovenian with English abstract)

BRUS, R. (2010): Growing evidence for the existence of glacial refugia of European beech (*Fagus sylvatica* L.) in the south-eastern Alps and north-western Dinaric Alps. Periodicum Biologorum, 112, 3: 239–246.

BRUS, R. – HORVAT-MAROLT, S. – PAULE, L. – GÖMÖRY, D. (1999): Genetska variabilnost bukve (*Fagus sylvatica* L.) v Sloveniji. [Genetic variability of European beech (*Fagus sylvatica* L.) in Slovenia]. Zbornik gozdarstva in lesarstva, 60: 85–106. (in Slovenian with English abstract)

BRUS, R. – HORVAT-MAROLT, S. – PAULE, L. (2000): Nova spoznanja o obstoju ledenodobnih zatočišč bukve (*Fagus sylvatica* L.) na ozemlju današnje Slovenije. [New recognitions of the existence of the beech (*Fagus sylvatica* L.) glacial refugia on the present Slovenian territory]. In: POTOČNIK, I. (ed.), Nova znanja v gozdarstvu - prispevek visokega šolstva : zbornik referatov študijskih dni, Kranjska gora, 11-12. 5. 2000. Biotehniška fakulteta, Oddelek za gozdarstvo in obnovljive gozdne vire, 77–88, Ljubljana. (in Slovenian with English abstract)

BRZEZIECKI, B. – KIENAST, F. – WILDI, O. (1995): Modelling potential impacts of climate change on the spatial distribution of zonal forest communities in Switzerland. Journal of Vegetation Science, 6: 257–268.

COMPS, B. – THIEBAUT, B. – SUGAR, I. – TRINAJSTIĆ, I. – PLAZIBAT, I. (1991): Genetic variation of the Croatian beech stands (*Fagus sylvatica* L.): spatial differentiation in conection with the environment. Annales des Sciences Forestiéres, 48: 15–28.

DAKSKOBLER, I. (2008): Pregled bukovih rastišč v Sloveniji. [A Review of beech sites in Slovenia]. Zbornik gozdarstva in lesarstva, 87: 3–14. (in Slovenian with English abstract)

ELLENBERG, H. (1996): Vegetation Mitteleuropas mit den Alpen, 5th edn.- Ulmer, Stuttgart, Germany.

FINKELDEY, R. (1993): Die Bedeutung allelischer Profile für die Konservierung genetischer Ressourcen bei Waldbäumen. Gött. Forstgenet. Bericht, 14, 176 p.

FOTELLI, N.M – RENNENBERG, H. – GEßLER, A. (2002): Effects of drought on the competitive interference of an early successional species (*Rubus fruticosus*) on *Fagus sylvatica* L. seedlings: $^{15}N$ uptake and partitioning, responses of amino acids and other N compounds. Plant Biol., 4: 311–320.

GEßLER, A. – KEITEL, C. – KREUZWIESER, J. – MATYSSEK, R. – SEILER, W. – RENNENBERG, H. (2007): Potential risks for European beech (*Fagus sylvatica* L.) in a changing climate. Trees, 21: 1–11.

GILLET, E.M. (1998): GSED – Geneti Structure from Electrophoresis Data, Version 1.1e. Institut für Forstgenetik und Forstpflanzenzüchtung Universität Gettingen.

GÖMÖRY, D. – PAULE, L. – BRUS, R. – ZHELEV, P. – TOMOVIĆ, Z. – GRAČAN, J. (1999): Genetic differentiation and phylogeny of beech on the Balkan Peninsula. J. Evol. Biol, 12: 746–754.

GREGORIUS, H.R. (1974): Genetischer Abstand zwischen Populationen. I. Zur Konzeption der genetischen Abstandsmessung. Silvae Genetica, 23: 22–27.

GREGORIUS, H.R. (1978): The concept of genetic diversity and differentiation. Theor. Appl. Genet. 74: 397–401.

GREGORIUS, H.R. – KRAUHAUSEN, J. – MÜLLER-STARCK, G. (1986): Spatial and temporal genetic differentiation among the seed in stand of *Fagus sylvatica* L. Heredity, 57: 255–262.

GREGORIUS, H.R. (1987): The relationship between the concepts of genetic diversity and differentiation. Teor. Appl. Genet., 74: 397–401.

KONNERT, M. (1995): Investigations on the genetic variation of beech (*Fagus sylvatica* L.) in Bavaria. Silvae Genetica, 44: 346–351.

KONNERT, M. – HENKEL, W. (1997): Untersuchungen zur genetischen Variation der Buche (*Fagus sylvatica* L.) in Thüringen. Allgemeine Forst-und Jagdzeitung, 168, 10: 128–190.

KONNERT, M. – HUSSENDOERFER, E. – DOUNAVI, A. (2004): Handbücher für Isoenzymanalyse. Anleitung für Isoenzymuntersuchung bei Buche (*Fagus sylvatica*), URL: (http://www.genre.de/fgrdeu/blag/iso-handbuecher).

KRAIGHER, H. – BOŽIČ, G. – VERLIČ, A. (2012): Seznam gozdnih semenskih objektov – stanje na dan 1. 1. 2012. [List of forest seed objects – State of art at the day 1. 1. 2012]. Urad. list Republike Slovenije, 13.1.2012, 3: 206–212.

KUTNAR, L. – ZUPANČIČ, M. – ROBIČ, D. – ZUPANČIČ, N. – ŽITNIK, S. – KRALJ, T. – TAVČAR, I. – DOLINAR, M. – ZRNEC, C. – KRAIGHER, H. (2002): Razmejitev provenienčnih območij gozdnih drevesnih vrst v Sloveniji na osnovi ekoloških regij. [The delimitation of the regions of provenance of forest tree species in Slovenia based on ecological regions]. Zbornik gozdarstva in lesarstva, 67: 73–117. (in Slovenian with English abstract)

KUTNAR, L. – KOBLER, A. (2011): Prediction of forest vegetation shift due to different climate-change scenarios in Slovenia. Šumarski list 135 (3–4): 113–126.

KUTNAR, L. – MATIJAŠIĆ, D. – PISEK, R. (2011): Conservation Status and Potential Threats to Natura 2000 Forest Habitats in Slovenia. Šumarski list, 135 (5–6): 215–231.

LÖCHELT, S. – FRANKE, A. (1995): Bestimmung der genetischen Konstitution von Buchen - Beständen (*Fagus sylvatica* L.) entlang eines Höhentransekten von Frieburg auf den Schauinsland. Silvae Genetica, 44, 312–318.

MAGRI, D. –,VENDRAMIN, G.G. – COMPS, B. – DUPANLOUP, I. – GEBUREK, T. – GÖMÖRY, D. – LATAŁOWA, M. – LITT, T. – PAULE, L. – ROURE, J.M. – TANTAU, I. – KNAAP, W.O. – PETIT, R.J. – BEAULIEU, J.L. (2006): A new scenario for the Quaternary history of European beech populations: Palaeobotanical evidence and genetic consequences. New Phytologist, 171, 1: 199–221.

MARINČEK, L. – ČARNI, A. (2002): Commentary to the vegetation map of forest communities of Slovenia in a scale 1:400.000. Založba ZRC, ZRC SAZU, Biološki inštitut Jovana Hadžija, Ljubljana, 158 p.

MÁTYÁS, CS. – BERKI, I. – CZÚCZ, B. – GÁLOS, B. – MÓRICZ, N. – RASZTOVITS, E. (2010): Future of Beech in Southeast Europe from the Perspective of Evolutionary Ecology. Acta Silv. Lign. Hung. 6: 91–110.

MÜLLER-STARCK, G. – BARADAT, PH. – BERGMANN, F. (1992): Genetic variation within European tree species. New Forests, 6: 23–47.

MÜLLER-STARK, G. – WÜHLISCH, G. VON – KONNERT, M. (2001): Labormanual zur Durchfuehrung von Isoenzymanalysen bei Buche (*Fagus sylvatica* L.), URL: (http://www.forst.tumuenchen.de/EXT/LST/GENET/index.htm).

NEI, M. (1973): Analysis of gene diversity in subdivided populations. Proc. Nat. Acad. Sci. USA, 70, 3321–3323.

OGRIS, N. – JURC, M. – JURC, D. (2008): Varstvo bukovih gozdov - danes in jutri. [Protection of beech forests – today and tomorrow]. In: BONČINA, A. (ed.). Bukovi gozdovi-ekologija in gospodarjenje: zbornik razširjenih povzetkov predavanj. Biotehniška fakulteta, Oddelek za gozdarstvo in obnovljive gozdne vire, Ljubljana, 36–39. (in Slovenian with English abstract)

OSOLE, F. (1990): Lukenjska jama pri Prečni : Novo mesto. [The Lukanja cave at Prečno : Novo mesto]. Arheo (Ljubljana), special number, 7–10. (in Slovenian)

PAULE, L. (1995): Gene conservation in European beech (*Fagus sylvatica* L.). Forest Genetics 2 (3): 161–170.

PAULE, L. – GÖMÖRY, D. – VYŠNY, J. (1995): Genetic diversity and differentiation of beech populations in Eastern Europe. In: MADSEN, S. (ed.). Genetics and silviculture of beech. Forskningsserien (Copenhagen) 11: 159–167.

PERKO, F. (2007): Slovenian forests and forestry. Ljubljana, Zveza gozdarskih društev, Ministrtvo za kmetijstvo, gozdarstvo in prehrano RS, Zavod za gozdove Slovenije, Ljubljana, 39 p.

PEUKE, A.D. – SCHRAML, C. – HARTUNG, W. – RENNENBERG, H. (2002): Identification of drought-sensitive beech ecotypes by physiological parameters. New Phytol., 154: 373–387.

ROBSON, T.M. – ALIÁ, R. – BOŽIČ, G. – CLARK, J. – FORSTREUTER, M. – GÖMÖRY, D. – LISEBACH, M. – MERTENS, P. – RASZTOVITS, E. – ZITOVÁ, M. – WÜHLISCH, G. VON (2011): The timing of leaf flush in European beech (*Fagus sylvatica* L.) saplings. In: Proceedings of the COST E52 Final Meeting "Evaluation of beech genetic resources for sustainable forestry", Spain, May 2010, Monografías INIA Forestal, 22: 61–79.

SANDER, T. – KÖNIG, S. – ROTHE, G.M. – JANSSEN, A. – WEISGERBER, H. (2000): Genetic variation of European beech (*Fagus sylvatica* L.) along an altitudinal transect at mount Vogelsberg in Hesse, Germany. Molecular Ecology, 9: 1349–1361.

SFS (2012): Forest Report for the year 2011. Slovenia Forest Service, Ljubljana, 133 p. (in Slovenian)

SIMONČIČ, P. – URBANČIČ, M. – BRIŠNIK, T. – FERLAN, M. – KOBAL, M. – VERLIČ, A. (2007): Introduction into project Carbon dynamics in natural beech forest. In: FERLAN, M. (ed.). Carbon dynamics in natural beech forests, Ljubljana: Slovenian Forestry Institute, 23 p.

SITTLER, B. (1981): Experimentelle ökologische Untersuchungen an 15 slowenischen Buchenprovenienzen zur Beurteilung ihrer Anbaufähigkeit in der Bundesrepublik Deutschland. [Experimental ecological studies on the origin of 15 Slovenian beech provenances to assess their growth in the Federal Republic of Germany]. Dissertation. Albert-Ludwig-University of Freiburg, 205 p.

ŠERCELJ, A. (1996): The origins and development of forest in Slovenia. Slovenska akademija znanosti in umetnosti. Razred za naravoslovne vede, Dela (Opera), Ljubljana, 35: 142 p.

URBANČIČ, M. – SIMONČIČ, P. – PRUS, T. – KUTNAR, L. (2005): Atlas gozdnih tal Slovenije. [Atlas of forest soils in Slovenia]. Ljubljana: Zveza gozdarskih društev Slovenije, Gozdarski vestnik, Gozdarski inštitut Slovenije, Ljubljana, 100 p. (in Slovenian with English abstract)

WÜHLISCH, G. VON (2008): EUFORGEN Technical Guidelines for genetic conservation and use for European beech (*Fagus sylvatica*). Bioversity International, Rome, 6 p.

VERSITA DOI: 10.2478/v10303-012-0007-2 Acta Silv. Lign. Hung., Vol. 8 (2012) 87–102

# The Role of Forests in Mitigating Climate Change – a Case Study for Europe

Borbála GÁLOS[a*] – Andreas HÄNSLER[b] – Georg KINDERMANN[c] – Diana RECHID[b] – Kevin SIECK[b] – Daniela JACOB[b]

[a] Institute of Environment and Earth Sciences, University of West Hungary, Sopron, Hungary
[b] Climate Service Center, Germany – eine Einrichtung am Helmholtz-Zentrum Geesthacht
[c] IIASA, International Institute for Applied Systems Analysis Austria

**Abstract** – A regional-scale case study has been carried out to assess the possible climatic benefits of forest cover increase in Europe. For the end of the 21st century (2071–2090) it has been investigated, whether the projected climate change could be reduced assuming potential afforestation of the continent. The magnitude of the biogeophysical effects of enhanced forest cover on temperature and precipitation means and extremes have been analyzed relative to the magnitude of the climate change signal applying the regional climate model REMO. The simulation results indicate that in the largest part of the temperate zone potential afforestation may reduce the projected climate change through cooler and moister conditions, thus could contribute to the mitigation of the projected climate change for the entire summer period. The largest relative effect of forest cover increase can be expected in northern Germany, Poland and Ukraine. Here, the projected precipitation decrease could be fully compensated, the temperature increase could be relieved by up to 0.5 °C, and the probability of extremely warm and dry days could be reduced. Results can help to identify the areas, where forest cover increase could be the most effective from climatic point of view. Thus they can build an important basis of the future adaptation strategies and forest policy.

**afforestation / biogeophysical feedbacks / climatic extremes / climate change mitigation**

**Kivonat – Esettanulmány az erdők klímavédelmi szerepének vizsgálatára Európában.** Az esettanulmány célja az erdőterület növekedés éghajlati hatásainak, a klímaváltozás mérsékelésében betöltött szerepének számszerűsítése Európában. A REMO regionális klímamodell segítségével vizsgáltuk, hogy a feltételezett potenciális erdőtelepítéssel milyen irányban és mértékben befolyásolhatók a 2071–2090-es időszakra előrevetített hőmérséklet- és csapadéktendenciák. A modellszimulációk eredményei alapján, potenciális erdőtelepítés feltételezésével nyáron a mérsékelt övi területek döntő része hűvösebb, csapadékosabb lehet. A legnagyobb hatás Németország és Lengyelország északi részén, valamint az ukrán-belorusz-orosz határvidéken várható. Ezeken a területeken az erdőtelepítés hatása a hőmérsékletre egy nagyságrenddel kisebb, mint az üvegházgáz koncentráció változásáé. A klímaváltozással járó csapadékmennyiség-csökkenés azonban szinte teljes egészében kiegyenlíthető lenne, és a szélsőségesen meleg és száraz napok gyakorisága csökkenhet. Az erdő-klíma kölcsönhatások számszerűsítése nem csak az erdők klímavédelmi szerepéről ad információt, hanem az éghajlatváltozás következményeinek megelőzését, enyhítését célzó stratégiák alapja is lehet.

**erdőtelepítés / felszín–légkör kölcsönhatás / klímaváltozás mérsékelő hatás / időjárási szélsőségek**

* Corresponding author: bgalos@emk.nyme.hu; H-9400 SOPRON, Bajcsy Zs. u. 4.

## 1 INTRODUCTION

Human activity influences climate through changes of the chemical composition of the atmosphere as well as through land cover and land use change. Climate conditions in the 21st century driven by emission change and its impacts on different spatial and temporal scales and sectors have been addressed by several national and international research projects in the last decade (Christensen et al. 2007, Jacob et al. 2008, van den Linden – Mitchell 2009). Regional climate projections for Europe agree in a robust warming tendency in all seasons towards the end of this century. The largest temperature increase is expected in the Mediterranean region, Southern France and over the Iberian Peninsula (Christensen – Christensen 2007). Here, the probability of extreme high temperatures, heat waves can also increase compared to the present day conditions (Beniston et al. 2007, Vidale et al. 2007, Beniston 2009, Fischer et al. 2010). The spatial patterns of the precipitation changes show an increase in northern Europe (especially in winter) and decrease in the southern regions (especially in the Mediterranean area in summer; Christensen – Christensen 2007). The Mediterranean and the South-East European regions are the most prone to higher risks of prolonged dry spells by 2100 (Beniston et al. 2007, Bartholy et al. 2007, Mishra – Singh 2010). Whereas in Northern to North-Eastern Europe the number of days with intense precipitation is very likely to increase, which can result in a rise in flood frequencies (Pal et al. 2004, Beniston et al. 2007, Buonomo et al. 2007). Changes of the climatic means and extremes already show impacts on the key sectors such as hydrological systems, infrastructure, human health, agriculture and forestry, which are expected to be more severe under future climate conditions.

Climate change can induce a shift of the spatial distribution of the major ecosystem types. For forests, this phenomenon occurs not only in the boreal zone (through the northern shift of the taiga-tundra limit) but also on the lower limit of the forest distribution (Berki et al. 2009, Mátyás et al. 2010, Czúcz et al. 2011, Führer et al. 2011). Here, ecological models of forest distribution expect growth decline and mass mortality of many zonal tree species whose distributions are limited primarily by recurrent droughts (Mátyás et al. 2010, Czúcz et al. 2011, Bredemeier 2011, Rasztovits et al. 2012). Land cover is also influenced by land management and land use policy. Land cover and land use and their changes are important drivers of weather and climate (Pielke et al. 1998, Drüszler et al. 2010), they can have a determining role in climate regulation. They affect the physical properties of the surface (biogeophysical processes and feedbacks) that control the water and energy exchange between land and atmosphere. Through chemical reactions (biogeochemical processes and feedbacks) they influence the terrestrial carbon exchange and the fluxes of other trace gases and aerosols (Pielke et al. 1998, Feddema et al. 2005, Pitman 2003). There are climate forcings that have basically a warming (e.g. greenhouse gases) or cooling effect (e.g. sulphate aerosols; Pielke et al. 2011). In contrast to them the sign of the temperature response to land cover and land use change are determined by various contrasting feedbacks. They can also depend on the type of the change as well as on the climatic, soil and vegetation characteristics of a region and the length of the analyzed time scale (Sánchez et al. 2007, Anav et al. 2010, Teuling et al. 2010, Wramneby et al. 2010).

Historical land cover and land use changes and their observed impacts on local and regional climate have been documented or reviewed by several studies for different countries (e.g. Feddema et al. 2005, Pielke et al. 2011, Pongratz et al. 2011). One of the major conclusions was that on regional scale, climate response of land cover and land use change has similar magnitude but opposite sign compared to the impact of elevated greenhouse gases and resulting changes of sea surface temperature and sea ice extent (de Noblet-Ducoudre et al. 2012).

For future time periods, changes of the land cover and land use can lead to the enhancement or reduction of the projected climate change signals expected from increased atmospheric greenhouse gas concentration (Feddema et al. 2005, Bonan 2008, Wramneby et al. 2010). Therefore understanding and assessment of their feedbacks on climate are essential for the adaptation and mitigation strategies.

Europe (with Russia) is the only continent with a significant increase of forest cover in recent times. In the last two decades the annual area of natural forestation and forest planting amounted to an average of 0.78 million hectares/year (FAO 2011). The climatic influence of these land cover changes is still poorly understood. The role of temperate forests in the climate change mitigation is considered smaller or uncertain compared to boreal and tropical forests (Bala et al. 2007, Bonan 2008, Jackson et al. 2008). Model results show contradictory results regarding to their net climate forcing and benefit (e.g. Anav et al. 2010, Gálos et al. 2011a,b).

One of the scientific goals of the EC-FP7 project CC-TAME (Climate Change – Terrestrial Adaptation and Mitigation in Europe) was to prepare fine-scale studies for the assessment of the climate protecting effects of land cover and land use change (e.g. Gálos – Jacob 2012). In order to contribute to this aim we prepared a regional scale case study to assess

- the biogeophysical effects of a potential afforestation in summer, for the end of the 21st century and its regional differences within Europe,
- the magnitude of the biogeophysical feedbacks of forest cover increase compared to the projected climate change signal with special focus on the probability and severity of temperature and precipitation extremes.

## 2 MODEL AND METHODS

### 2.1 The regional climate model REMO

The climate change driven by emission change and land cover change have been studied applying the REgional climate MOdel, REMO (Jacob et al. 2001, Jacob et al. 2007). This is a regional three-dimensional numerical model of the atmosphere. The prognostic variables are calculated based on the hydrostatic approximation. Land cover is described by its physical properties in REMO: leaf area index and fractional vegetation cover for the growing and dormancy season, background albedo, surface roughness length of the vegetation, forest ratio, plant-available soil water holding capacity and volumetric wilting point. These properties are allocated in the global dataset of land surface parameters (Hagemann et al. 1999, Hagemann 2002) for each land cover type. Vegetation phenology is represented by the mean climatology of the annual cycle of leaf area index, vegetation ratio and background albedo (Rechid and Jacob 2006, Rechid et al. 2008a,b). The values of these vegetation characteristics are varying monthly throughout the year, the other land surface parameters remain constant in time. In the current model version biogeochemical processes and vegetation dynamics are not considered.
For Europe, REMO has been validated against observations and the simulation results have been compared to an ensemble of regional climate model projections (Déqué et al. 2005).

### 2.2 Experimental setup

The simulations have been carried out for Europe (*Figure 1*), with 0.22° horizontal grid resolution. REMO was driven with lateral boundary conditions from the coupled atmosphere–ocean GCM ECHAM5/MPI-OM (Roeckner et al. 2006, Jungclaus et al. 2006).

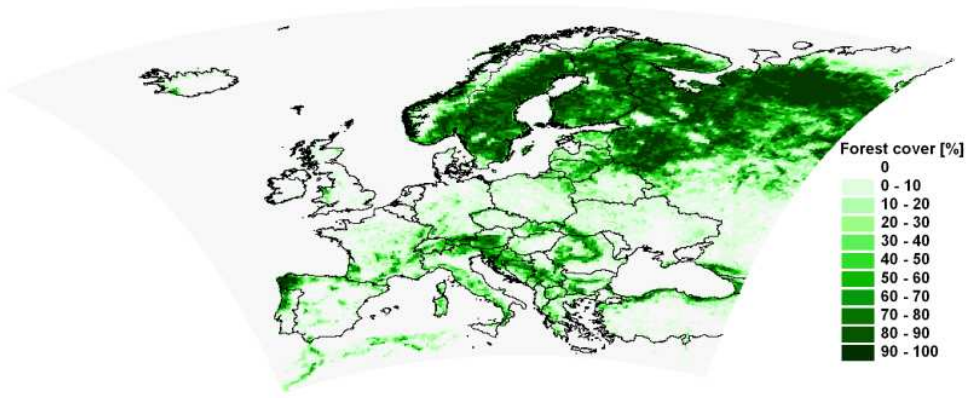



*Figure 1. Simulation domain with the present (unchanged) forest cover in the model. Horizontal resolution: 0.22 °*

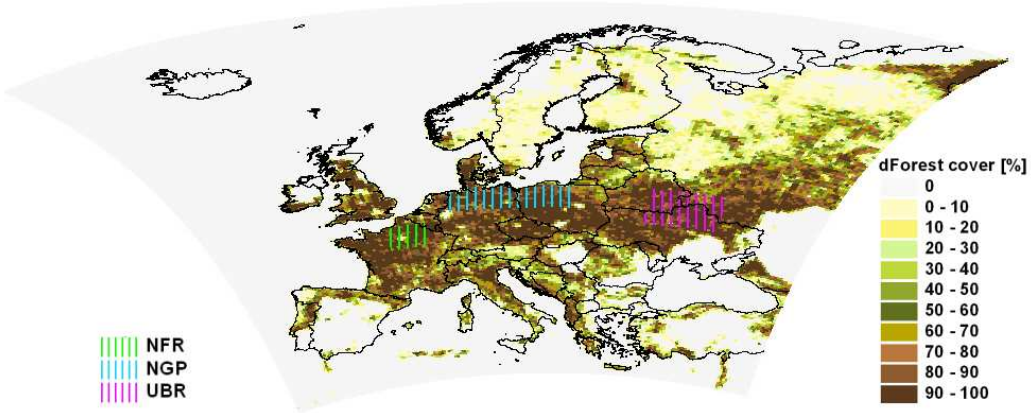



*Figure 2. Increase of the forest cover in the potential afforestation simulation compared to the present (unchanged) forested area in the model. The three sub-regions are marked (NFR: Northern France; NGP: Northern Germany and Poland; UBR: border region of Ukraine, Belarus and Russia)*

The following experiments have been performed and analysed (*Table 1*):

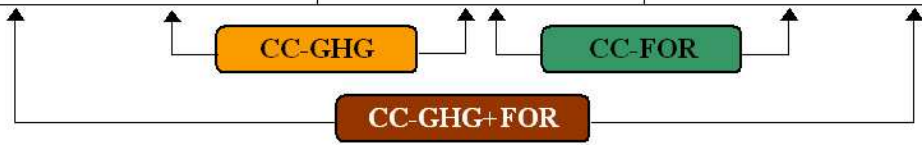
- Reference simulation for the past (1971–1990) with present (unchanged) forest cover.
- Emission scenario simulations for the future (2071–2090) with unchanged forest cover applying the A2 IPCC-SRES emission scenario (continuously increasing global population and regionally oriented economic growth that is more fragmented and slower than in other storylines; Nakicenovic et al. 2000). This served as reference simulation for the land cover change experiments.
  Emission scenario simulation with potential afforestation for 2071–2090. The potential afforestation map (*Figure 2*) is based on the net primary production map for Europe derived from remotely sensed MODIS (Moderate-Resolution Imaging Spectroradiometer) products, precipitation and temperature conditions from the Wordclim database and soil conditions from the International Institute for Applied Systems Analysis. In our study additional forested areas were assumed to be deciduous, because for the end of the 21st century, projected climate conditions will not be sufficient for larger continuous coniferous forest blocks in the temperate and Mediterranean regions.

The new potential forest cover map has been included into REMO by modification of all characteristic land surface parameters (i.e. leaf area index and fractional vegetation

cover for the growing and dormancy season, background albedo, surface roughness length due to vegetation, forest ratio, plant-available soil water holding capacity and volumetric wilting point).

*Table 1. Analyzed data and time periods (CC-GHG: climate change driven by change of the greenhouse gas concentration, CC-FOR: climate change driven by potential afforestation CC-GHG+FOR: climate change driven by change of the greenhouse gas concentration and potential afforestation)*

| Experiment | Reference simulation | Emission scenario simulation | |
|---|---|---|---|
| Time period | 1971-1990 | 2071-2090 | |
| Greenhouse gas forcing | | IPCC-SRES emission scenario A2 | |
| Land cover characteristics | Present forest cover unchanged | Present forest cover unchanged | Deciduous forests cover all additional vegetated area |




### 2.3 Method of analyses

The analyses of the simulation results focused on the summer months (June, July, August), because of the high radiation input, intense heat and mass exchange.

*Climate change driven by change of the greenhouse gas (GHG) concentration* (CC-GHG; *Table 1*) has been investigated comparing the summer precipitation sums and 2m-temperature means for the period 2071–2090 (without any land cover changes) to 1971–1990. *Climate change driven by potential afforestation* (CC-FOR; *Table 1*) has been analyzed comparing the simulation results with- and without forest cover increase for the future time period (2071-2090). *Climate change driven by change of the greenhouse gas concentration and potential afforestation* (CC-GHG+FOR; *Table 1*) has been assessed comparing the results of the potential afforestation experiment (2071–2090) to the reference study in the past (1971-1990). The sign and the magnitude of climatic effects of potential afforestation have been analyzed relative to the climate change signal, and the regional differences have been determined within Europe.

Three sub-regions have been selected, where a large forest cover increase has been assumed in the case study and based on the simulation results forests can play a major role in altering the climate change signal. Here, biogeophysical effects of potential afforestation on temperature and precipitation means and extremes have been analyzed more in detail. The sub-regions are the following: Northern France (NFR), Northern Germany and Poland (NGP) and the border region of Ukraine, Belarus and Russia (UBR; *Figure 2*).

## 3 RESULTS

### 3.1 Effects of change of the greenhouse gas concentration and potential afforestation on the summer temperature mean

First, change of the summer temperature without any forest cover change has been analyzed for the end of the 21st century (2071–2090) with reference to the 20-year period in the 20th century (1971–1990). Corresponding to the results of other regional climate simulations for

Europe, REMO projects a significant increase of the summer temperature mean, which is expected to be the largest in the southern part of the continent and in the north-eastern areas of the simulation domain (*Figure 3a*). In the case of potential afforestation, the higher leaf area index and roughness lengths of forests support the enhanced ability of evapotranspiration (as long as there is enough moisture available in the soil), which leads to cooler conditions in most parts of the temperate zone. In the northern part of Central Europe and in North Ukraine, summer temperature mean can be 0.5 °C lower compared to the reference simulation (*Figure 3b*). Portugal, the Mediterranean coasts and the southern part of the boreal zone show a shift into the warmer direction. For the Mediterranean region a possible reason for it can be that in this dry area vegetation has deeper roots in the reference simulation than forests in the potential afforestation experiment. It means in the model that less water is available for cooling through evapotranspiration.

*Figure 3c* represents the combined effect of the GHG concentration change and potential afforestation on the summer temperature mean. In the temperate zone afforestation can reduce the projected warming. However, the temperature change signal for potential afforestation is smaller than for GHG concentration change in the whole continent (*Figure 3c*) thus cannot offset the robust warming. Increase of the forest cover may enhance the climate change signal in the boreal and in the Mediterranean regions but its magnitude is relatively small compared to the effect of the GHG concentration changes.

### 3.2 Effects of change of the greenhouse gas concentration and potential afforestation on the summer precipitation sum

For the end of the 21st century, an increase of temperature is projected to occur with a decrease of precipitation in South and Central Europe and in the southern part of Scandinavia, whereas Northeast Europe can be characterized by warmer and wetter conditions (*Figure 4a*). Assuming potential afforestation, higher amount of summer precipitation is expected in the largest part of the temperate zone, especially in the northern areas. Here, precipitation increase can reach 50–75 mm in summer mean (*Figure 4b*). Enhanced forested area resulted in less precipitation in the northern part of Scandinavia and Russia as well as in smaller areas in Spain and in Central and Southeast Europe (*Figure 4b*). In contrast with temperature, precipitation change cannot be directly correlated with the local forest cover change, because its formation is influenced also by large-scale circulation.

Increase of forest cover can amplify the projected precipitation change in Sweden, Belarus and Southwest Russia (*Figure 4c*). Whereas in the northern part of central Europe, Ukraine and eastern Finland the precipitation change driven by GHG concentration change has the opposite sign than due to potential afforestation (*Figures 4a-b*). Thus depending on their magnitude, afforestation can reduce or fully compensate the effects of the GHG concentration change. *Figure 4c* shows the areas, where precipitation change driven by afforestation is the same or even larger in magnitude than the climate change signal.

Summing up the results for summer in the period 2071–2090, the cooling and moistening effects of potential afforestation are dominant in most parts of the temperate zone. For temperature, approximately 15–20% of the climate change signal can be relieved by forest cover increase. Whereas for precipitation, climatic effects of potential afforestation and GHG concentration change have the same order of magnitude but opposite sign. The south-western part of Europe is the most affected by warming and drying driven by GHG concentration change. Here, climatic benefits of forests are simulated to be weaker.

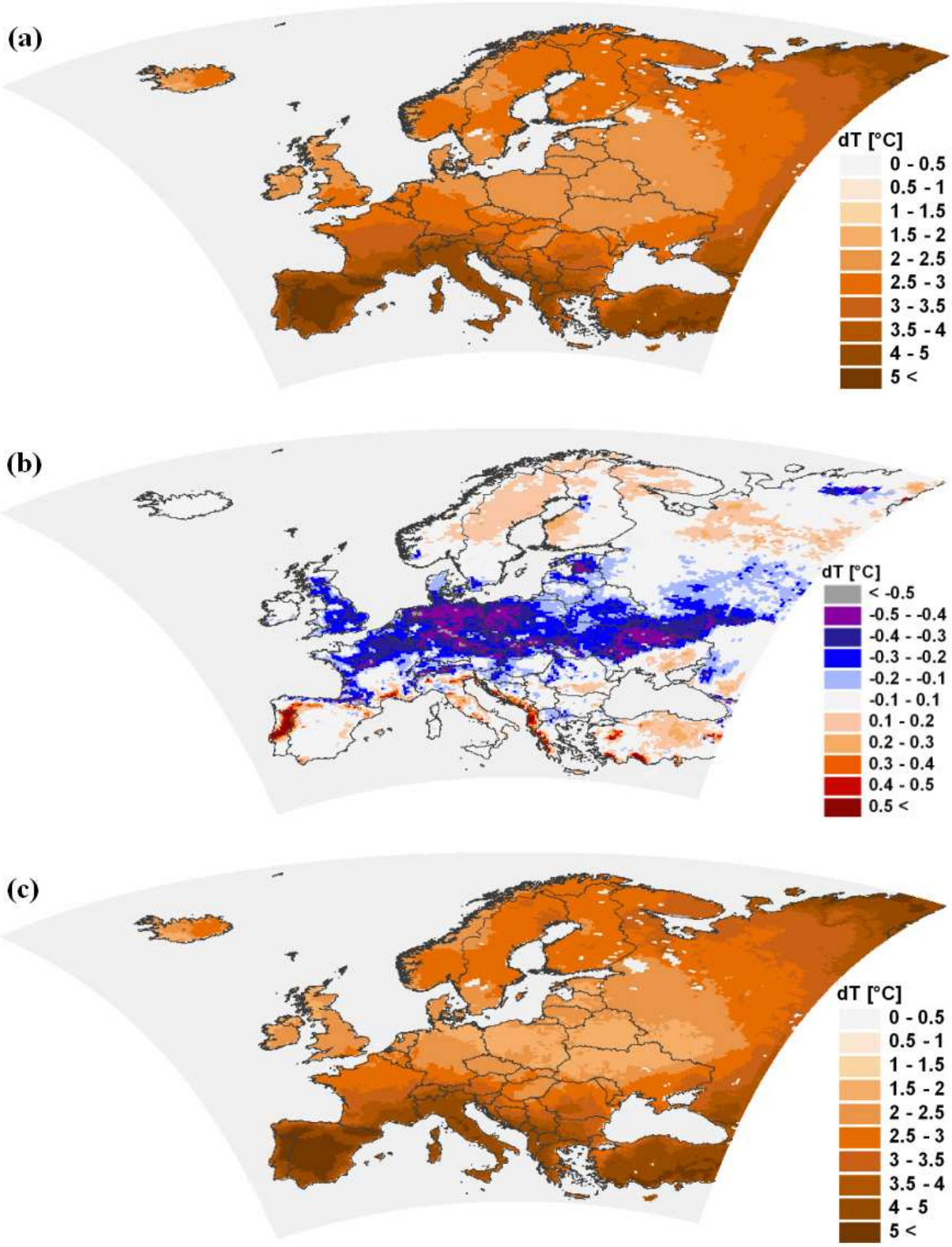



*Figure 3. Change of mean summer temperature (dT) driven by (a) change of the GHG concentration (2071–2090 vs. 1971–1990) (b) potential afforestation (2071–2090) and (c) GHG concentration change + potential afforestation (2071–2090 vs. 1971–1990).*

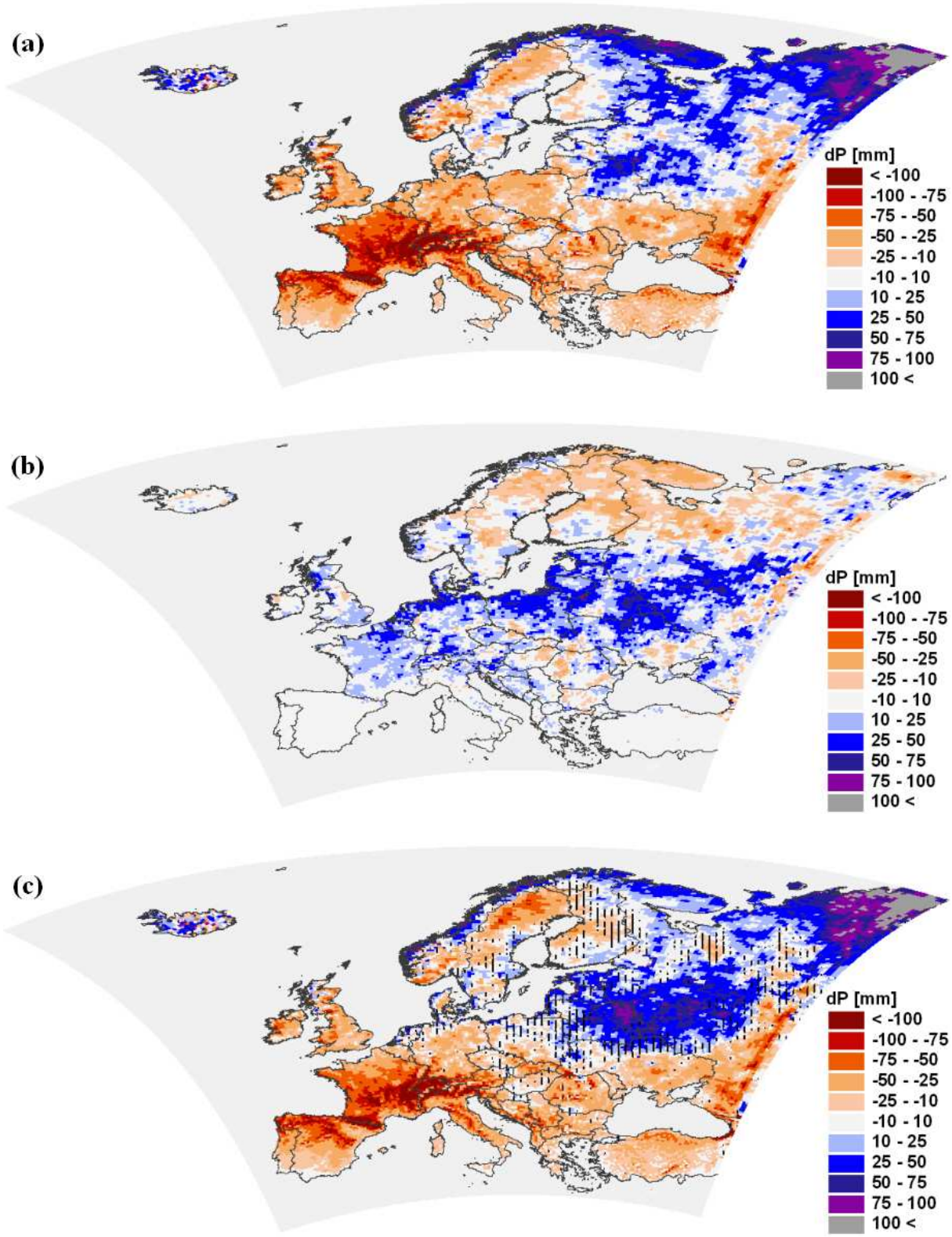



*Figure 4. Change of summer precipitation sum (dP) driven by (a) change of the GHG concentration (2071–2090 vs. 1971–1990) (b) potential afforestation (2071–2090) and (c) GHG concentration change + potential afforestation (2071–2090 vs. 1971–1990). The regions are marked, where precipitation change driven by afforestation is the same or larger in magnitude than the climate change signal (only effects over land are shown)*

## 3.3 Sign and magnitude of the climate change altering effects of forest cover increase in selected regions

### *3.3.1 Temperature and precipitation means*

In *Sect. 3.2* it has been shown that in the northern part of central Europe and Ukraine forests can play a major role in altering the climate change signal. Three sub-regions have been selected within these areas to analyse the temperature and precipitation conditions more in detail (NFR: Northern France, NGP: Northern Germany and Poland, UBR: the border region of Ukraine, Belarus and Russia; *Figure 2*).

*Figure 4a-b* shows that for both investigated climatic variables the effects of GHG concentration change can be reduced by the forest cover increase. The temperature change signals for potential afforestation (–0.3 – –0.4 °C) are weaker than for GHG concentration change (+2.1 – +2.4 °C; *Figure 5*). In smaller areas, climate change mitigating effect of potential afforestation can be stronger (Gálos – Jacob, 2012). Based on the A2 emission scenario, 28 % precipitation decrease (–66 mm) is expected in the Northern French area. One third of this amount (22 mm) could be reduced by forest cover increase. In the region of Northern Germany and Poland, the projected drying is smaller (–13 %; –35 mm). If GHG concentration change occurred together with potential afforestation, 80 % of the original climate change signal could be relieved (*Figure 5*). In the Ukrainian border region, increase of summer precipitation sum driven by potential afforestation (+14 %; +34 mm) can exceed the very small drying of the area due to the enhanced GHG concentration (*Figure 5*). Thus afforestation would fully compensate the projected climate change signal, as long as there is enough soil moisture available. The combined effect of forest cover and GHG concentration changes for 2071–2090 would result in a net precipitation increase compared to the reference simulation for the past (1971–1990) without any land cover change.

For the analyzed variables, the magnitude of impacts of forest cover increase is similar in the tree sub-regions. Consequently, the regional differences in the relative climatic effects of potential afforestation can be explained by the differences in the temperature and precipitation signal driven by GHG concentration change.

### *3.3.2 Temperature and precipitation extremes*

Distribution of the daily temperature means are shifted towards the warmer direction under future climate conditions for the NGP region (*Figure 6*). *Figure 6* illustrates that in the potential afforestation case study the value of the 95th percentile is 0.5 °C lower than in the emission scenario simulation with unchanged land cover. Consequently, increase of the forest cover may contribute to the decrease of the probability and severity of extreme high temperatures, thereby to the reduction of the projected climate change signal. (The NFR and UBR regions show similar effects; not shown).

In each of the selected regions the probability of summer days (Tmax > 25 °C), hot days (Tmax > 30 °C) and extremely hot days (Tmax > 35 °C) are projected to increase significantly towards the end of the 21st century. Changes driven by potential afforestation have the opposite sign but they are relatively small compared to the effect of the GHG concentration changes (not shown).

Under future GHG concentrations the total number of dry days (daily precipitation sum < 1 mm) is projected to be significantly larger. In case of potential afforestation around half of this signal could be relieved in the NGP and UBR regions (*Table 2*). The probability of extremely large daily precipitation amounts (larger than 95th percentile) is simulated to become higher assuming forest cover increase for all of the three regions. In the UBR area, potential afforestation may enhance the effects of GHG concentration change, resulting in almost twice as much severe precipitation events than with unchanged vegetation (*Table 2*).

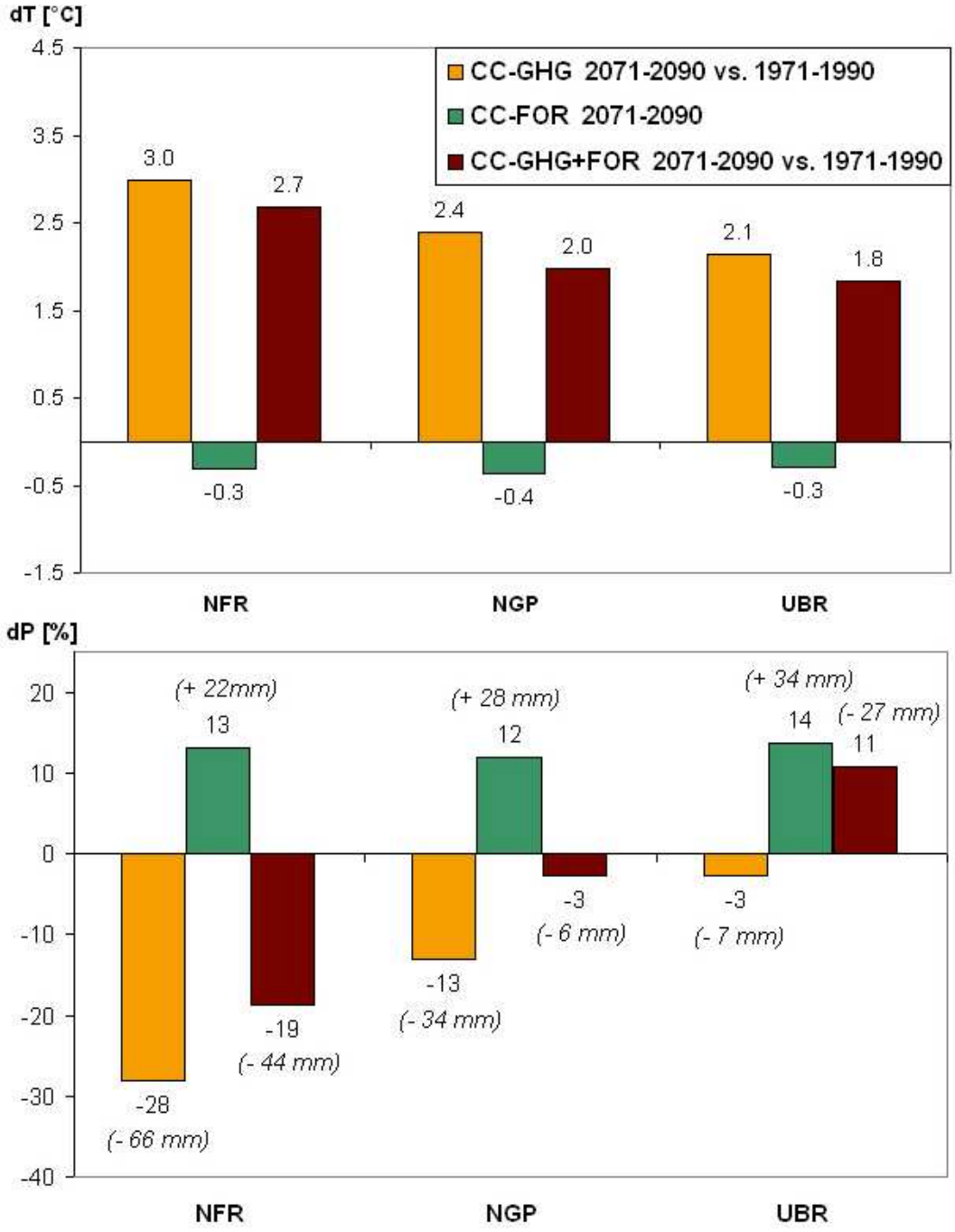



*Figure 5. Change of the summer temperature mean (dT; top) and precipitation sum (dP; bottom) driven by change of the GHG concentration (CC-GHG; 2071–2100 vs. 1971–1990), potential afforestation (CC-FOR; 2071–2090) and GHG concentration change + potential afforestation (CC-GHG+FOR; 2071–2090 vs. 1971–1990) in the three investigated regions (NFR: Northern France; NGP: Northern Germany and Poland; UBR: border region of Ukraine, Belarus and Russia)*

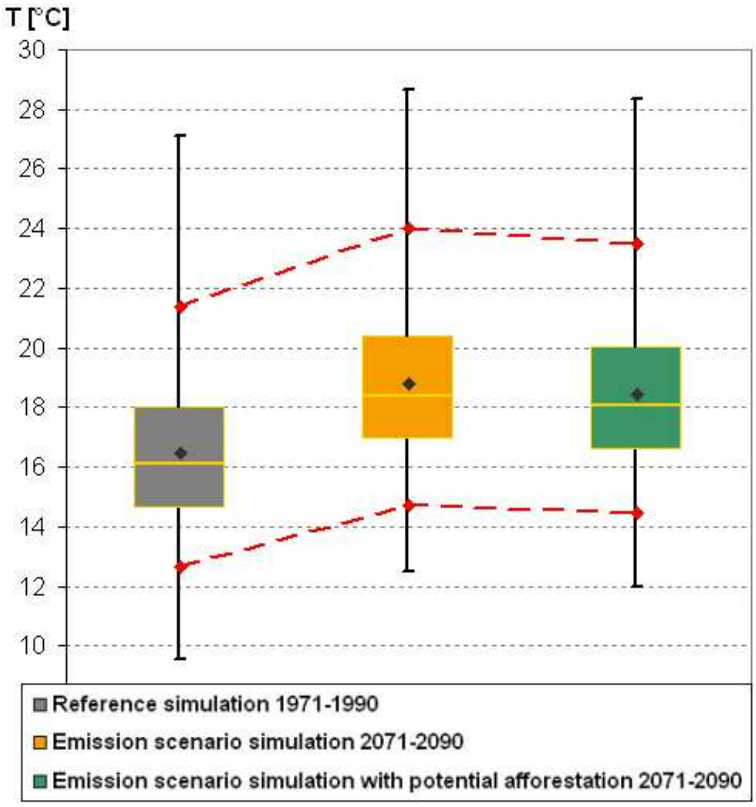



*Figure 6. Daily mean temperatures (T) in the summer months within the investigated 20-year time periods for the region Northern Germany and Poland (NGP). The bottom and the top of the boxes are representing the 25th and 75th percentile values. The end of the bars corresponds to the minimum and maximum of the data. The change of the 5th and 95th percentile values driven by GHG concentration change and potential afforestation is visualized by dashed lines*

*Table 2. Total number of daily precipitation extremes for summer in the investigated 20-year time periods. (CC-GHG: climate change driven by change of the greenhouse gas concentration, CC-FOR: climate change driven by potential afforestation, CC-GHG+FOR: climate change driven by change of the greenhouse gas concentration and potential afforestation; Rday: daily precipitation sum; R95%: 95th percentile of the daily precipitation in the period 1971–1990; NFR: Northern France; NGP: Northern Germany and Poland; UBR: border region of Ukraine, Belarus and Russia.)*

| Extreme index | Definition [unit] | Region | Number of days (1971–1990) | Change of the number of days | | |
|---|---|---|---|---|---|---|
| | | | | **CC-GHG** | **CC-FOR** | **CC-GHG+FOR** |
| Number of dry days | Rday < 1 mm [day] | NFR | 909 | +234 | –55 | +179 |
| | | NGP | 564 | +178 | –88 | +90 |
| | | UBR | 712 | +73 | –42 | +31 |
| Number of very wet days | Rday > R95% [day] | NFR | 47 | –11 | +13 | +2 |
| | | NGP | 64 | –5 | +15 | +10 |
| | | UBR | 57 | **+29** | **+26** | **+55** |

## 4 CONCLUSIONS

A regional scale case study has been carried out to investigate the role of the forest cover increase in adapting to climate change in Europe. For the end of the 21st century, simulation results of the regional climate model REMO have been analyzed in order to quantify the magnitude of the climatic effects of afforestation relative to the climatic impacts of GHG concentration change. The regions have been determined, in which forests can play a major role in altering the precipitation and temperature projections for summer. Here, the biogeophysical feedbacks of afforestation on the climatic means and extremes have been investigated more in detail.

Results of the sensitivity study can be summarized as follows:

- In the largest part of the temperate zone potential afforestation may reduce the projected climate change through cooler and moister conditions, thus could contribute to the mitigation of the projected climate change for the entire summer period, as long as there is enough soil moisture available.
- The largest relative effect of forest cover increase can be expected in northern Germany, Poland and Ukraine. Here, for temperature, approximately 15–20% (up to 0.4–0.5 °C) of the climate change signal can be relieved. For precipitation, climatic effects of potential afforestation and GHG concentration change have the same order of magnitude but opposite sign, which means, that projected precipitation decrease could be strongly reduced or fully compensated.
- In each of the analyzed sub-regions the strong increase of the total number of warm extremes can be slightly reduced by potential afforestation. Increase of forest cover would result in more severe heavy precipitation events. The probability of dry days would decrease.

The simulation results indicate that large, continuous forest blocks have robust biogeophysical effects on the regional climate. The magnitude of the climatic effects of afforestation relative to the climatic impact of GHG concentration change shows large spatial differences within Europe. Similarly to the conclusions of a Hungarian case study (Gálos et al. 2011a,b) there are regions, where increase of forest cover can play an important role in reducing the probability and severity of climatic extremes. Thus from a practical point of view, our assessment concerning to the climate change altering effects of forest cover change contributes to the future adaptation strategies in the European agriculture, forestry and water management. Our results also underline that in the mostly climate change affected part of the continent, vegetation feedbacks have weaker influence on the atmospheric circulation in comparison to the greenhouse-gas forcing (Betts 2007, Wramneby et al. 2010) and afforestation is not a substitute for reduced greenhouse-gas emissions (Arora – Montenegro, 2011). On local scale the benefits and ecological services of forest cover are highly valued.

For the introduced sensitivity study, one regional climate model has been applied driven by one emission scenario. The simulated impacts can also depend on the representation of the land surface properties and land cover related processes in the model (Boisier et al. 2012). There are differences among regional climate models in the parameters describing the land surface types and the role of these parameters in the vegetation-climate interactions that determines the sensitivity of the model to land cover changes. As example, in the land cover change experiment of Anav et al. (2010) the simulated total evapotranspiration shows large sensitivity to the modification of the stomatal resistance, which had a significant impact on the final conclusion of their study. In contrast to this, in the feedback-chain of the present study with REMO, the simulated transpiration reacts to the variation of the stomatal resistance, however the sensitivity of the total evapotranspiration is smaller because of the important role of the atmospheric demand. This can give an explanation to the contradictory

results of Anav et al. (2010) regarding to the impact of afforestation on temperature extremes for the same region. Further difference among regional climate models can be the number of the soil layers. The distribution of water in the soil and the water movement between the layers can also play an important role in the evapotranspiration through the available amount of soil moisture. The current bucket scheme in REMO is under improvement (Hagemann pers. comm.).

Further challenge is the elimination of the model-depending effects and reduction of the related uncertainties. To achieve this, multimodel ensembles of climate model simulations and intercomparison studies are essential, which is the aim of recent EU-projects (e.g. LUCID; de Noblet-Ducoudre et al. 2012). Land cover and land use changes directly affect regional climate. Increase of spatial resolution makes the correct description of land cover and the simulation of land – atmosphere feedbacks even more important. The land cover parameterization and land surface schemes of climate models should be validated and improved based on appropriate observational and satellite database, which is one of the research questions of the on-going EU project FLUXNET.

In our simulations, projected forest cover and forest composition shifts triggered by climate change (i.e. the expected reduction of the forested area and mass mortality in the drought threatened areas; Mátyás et al. 2010, Czúcz et al. 2011) have not been taken into account. There is no information available about the climate change effects on the distribution of forests as well as about the land use change due to land management and policy in Europe beyond limited case studies, so far. Vegetation dynamics and phenology were not considered, however, they can influence the simulated water and energy exchange between vegetation and atmosphere through the temporal change of the vegetation properties (e.g. albedo, leaf area index, surface roughness).

In these analyses we focused on the biogeophysical processes. But it is important to recognize that they can be intensified or dampened by the biogeochemical effects (e.g. carbon sequestration of forests and soil, which is one of the large unknowns under future climate conditions; Barcza et al. 2009, Booth et al. 2012). Higher $CO_2$ concentrations can also lead to the increase of the stomatal resistance thereby to the inhibition of the transpiration, which can amplify the global warming (Cao et al. 2010, Gopalakrishnan et al. 2011). Therefore to draw appropriate conclusions for decision makers about the role of the forests in the climate change mitigation and adaptation, the combined effects should be analyzed.

The accurate representation of land use and land cover change in past, present and future climate simulations is crucial to understand and quantify the interactions and feedbacks with the climate and socio-economic systems, respectively (Hibbard et al 2010, Mahmood et al. 2010, Pielke et al. 2011), as well as the ecosystem services of the land cover types and the role of humans in the climate system. These can be reached only by improved international and interdisciplinary collaboration across modelling, observing and measurement communities.

**Acknowledgements:** The climate model simulations have been carried out in the Max Planck Institute for Meteorology (MPI-M), Hamburg. The authors give special thanks to Swantje Preuschmann, Claas Teichmann and Christopher Moseley for their scientific contribution to the project that was an important basis of our study. Thanks to the Regional Modelling Group of the MPI-M and to Stefan Hagemann for the fruitful scientific discussions about the simulation results. We thank Prof. Dr. Csaba Mátyás for his expertise and suggestions regarding to the practical importance of this topic. The REMO simulations without land cover change have been carried out in the frame of the EU-project ENSEMBLES. This research was financially supported by the EC-FP7 project CC-TAME (www.cctame.eu; grant agreement n° 212535), as well as by the TÁMOP 4.2.2-08/1-2008-0020 and 4.2.2.B-10/1-2010-0018 "Talentum" joint EU-national research projects.

## REFERENCES

ANAV, A. – RUTI, P.M. – ARTALE, V. – VALENTINI, R. (2010): Modelling the effects of land-cover changes on surface climate in the Mediterranean region. Clim. Res. 41: 91–104.

ARORA, V.K. – MONTENEGRO, A. (2011): Small temperature benefits provided by realistic afforestation efforts. Nature Geoscience doi: 10.1038/NGEO1182,

BALA, G. – CALDEIRA, K. – WICKETT, M. – PHILLIPS, TJ. – LOBELL, D.B. – DELIRE, C. – MIRIN, A. (2007): Combined climate and carbon-cycle effects of large-scale deforestation. Proc Natl Acad Sci USA, 104: 6550–6555.

BARCZA, Z. – HASZPRA, L. – SOMOGYI, Z. – HIDY, D. – LOVAS, K. – CHURKINA, G. – HORVÁTH L. (2009): Estimation of the biospheric carbon dioxide balance of Hungary using the BIOME-BGC model. Időjárás 113(3): 203–209.

BARTHOLY, J. – PONGRÁCZ, R. – GELYBÓ, GY. (2007): Regional climate change expected in Hungary for 2071–2100. Applied Ecology and Environmental Research 5: 1–17.

BENISTON, M. – STEPHENSON, D.B. – CHRISTENSEN, O.B. – FERRO, C.A.T. – FREI, C. – GOYETTE, S. – HALSNAES, K. – HOLT, T. – JYLHÄ, K. – KOFFI, B. – PALUTIKOF, J. – SCHÖLL, R. – SEMMLER, T. – WOTH, K. (2007): Future extreme events in European climate: an exploration of regional climate model projections. Clim Change 81:71–95.

BENISTON, M. (2009): Trends in joint quantiles of temperature and precipitation in Europe since 1901 and projected for 2100. Geophys Res Lett 2009, 36:L07707. doi:10.1029/2008GL037119

BERKI, I. – RASZTOVITS, E. – MÓRICZ, N. – MÁTYÁS, CS. (2009): Determination of the drought tolerance limit of beech forests and forecasting their future distribution in Hungary. Cereal Research Communations 37: 613–616.

BETTS, R. (2007): Implications of land ecosystem-atmosphere interactions for strategies for climate change adaptation and mitigation. Tellus 59B: 602–615.

BOISIER, J.P. – DE NOBLET-DUCOUDRÉ, N. – PITMAN, A.J. – CRUZ, F.T. – DELIRE, C. –VAN DEN HURK, B.J.J.M. – VAN DER MOLEN, M.K. – MÜLLER, C. – VOLDOIRE, A. (2012): Attributing the impacts of land-cover changes in temperate regions on surface temperature and heat fluxes to specific causes: Results from the first LUCID set of simulations. J. Geophys. Res. 117: D12116, doi:10.1029/2011JD017106.

BOOTH, B.B.B. – CHRIS, D.J. – COLLINS, M. – TOTTERDELL, I.J. – COX, P.M – SITCH, S. – HUNTINGFORD, C. – BETTS, R.A.– HARRIS, G.R. – LLOYD, J. (2012): High sensitivity of future global warming to land carbon cycle processes. Environ. Res. Lett. 7 024002 (8pp) doi:10.1088/1748-9326/7/2/024002

BONAN, G.B. (2008): Forests and climate change: forcings, feedbacks, and the climate benefits of forests. Science 320: 1444–1449.

BREDEMEIER, M. (2011): Forest, climate and water issues in Europe. Ecohydrol 4:159–167.

BUONOMO, E. – JONES, R. – HUNTINGFORD, C. – HANNAFORD, J. (2007): On the robustness of changes in extreme precipitation over Europe from two high-resolution climate change simulations. Q J R Meteor Soc, 133:65–81.

CAO, L, – BALA, G. – CALDEIRA, K. – NEMANI, R. – BAN-WEISSA, G. (2010): Importance of carbon dioxide physiological forcing to future climate change. Proc. Natl. Acad. Sci. 107(21):9513–1918

CHRISTENSEN, J.H. – CARTER, T.R. – RUMMUKAINEN, M. – AMANATIDIS, G. (2007): Evaluating the performance and utility of regional climate models: the PRUDENCE project. Clim. Change 81: 1–6. doi:10.1007/s10584-006-9211-6

CHRISTENSEN, J.H. – CHRISTENSEN, O.B. (2007): A summary of the PRUDENCE model projections of changes in European climate by the end of this century. Clim. Change 81:7–30.

CZÚCZ, B. – GÁLHIDY, L. – MÁTYÁS, CS. (2011): Present and forecasted xeric climatic limits of beech and sessile oak distribution at low altitudes in Central Europe. Ann. For. Sci. 68 (1): 99–108.

DÉQUÉ, M. – JONES, R.G. – WILD, M. – GIORGI, F. – CHRISTENSEN, J.H. – HASSELL, D.C. – VIDALE, P.L. – ROCKEL, B. – JACOB, D. – KJELLSTRÖM, E. – DE CASTRO, M. – KUCHARSKI, F. VAN DEN HURK ,B. (2005): Global high resolution versus Limited Area Model scenarios over Europe: results from the PRUDENCE project. Climate Dynamics 25: 653–670.

DRÜSZLER, Á. – CSIRMAZ, K. – VIG, P. – MIKA, J. (2010): Effects of documented land use changes on temperature and humidity regime in Hungary. In: Saikia, S.P. (ed.): Climate Change. Dehradun: International Book Distributors. ISBN: 81–7089–370–4, 394–418.

FAO (2011): The State of the World's Forests 2011 Food and Agriculture Organisation of the United Nations; Rome

FEDDEMA, J.J. – OLESON, K.W. – BONAN, G.B. – MEARNS, L.O. – BUJA, L.E. – MEEHL, G.A. – WASHINGTON, W.M. (2005): The Importance of Land-Cover Change in Simulating Future Climates. Science 310: 1674 –1678.

FISCHER, E.M. – SCHÄR, C. (2010): Consistent geographical patterns of changes in high-impact European heatwaves. Nature Geoscience 2010, 3:398–403.

FÜHRER, E. – MAROSI, GY. – JAGODICS, A. – JUHÁSZ, I. (2011): A possible effect of climate change in forest management (in Hungarian). Erdészettudományi Közlemények 1(1): 17–28.

GÁLOS, B. – MÁTYÁS, CS. – JACOB, D. (2011a): Regional characteristics of climate change altering effects of afforestation. Environ. Res. Lett. 6 044010 (9pp) doi:10.1088/1748-9326/6/4/044010

GÁLOS, B. – JACOB, D. – MÁTYÁS, CS. (2011b): Effects of simulated forest cover change on projected climate change – a case study of Hungary. Acta Silv. Lign. Hung. 7: 49–62.

GÁLOS, B. – JACOB, D. (2012): Regional-scale assessment of the climatic role of forests under future climate conditions. In: Liu, G. (ed). Greenhouse Gases - Emission, Measurement and Management. ISBN 979-953-307-224-0, 295–314.

GOPALAKRISHNAN, R. – BALA, G. – JAYARAMAN, M. – CAO, L. – NEMANI, R.N.H. – RAVINDRANATH, N.H. (2011): Sensitivity of terrestrial water and energy budgets to CO2-physiological forcing: an investigation using an offline land model. Environ. Res. Lett. 6 044013 (7pp). doi:10.1088/1748-9326/6/4/044013

HAGEMANN, S. (2002): An improved land surface parameter dataset for global and regional climate models. Report 336, Max-Planck-Institute for Meteorology, Hamburg

HAGEMANN, S. – BOTZET, M. – DÜMENIL, L. – MACHENHAUER, M. (1999): Derivation of global GCM boundary conditions from 1 km land use satellite data. Report 289 Max-Planck-Institute for Meteorology, Hamburg

HIBBARD, K. – JANETOS, A. – VAN VUUREN, D.P. – PONGRATZ, J. – ROSE, S.K. – BETTS, R. – HEROLD, M. – FEDDEMA, J.J. (2010): Research priorities in land use and land cover change for the earth system and integrated assessment modelling. Int J Climatol, 30: 2118–2128.

JACKSON, R.B. – RANDERSON, J.T. – CANADELL, J.G. – ANDERSON, R.G. – AVISSAR, R. – BALDOCCHI, D.D. – BONAN, G.B. – CALDEIRA, K. – DIFFENBAUGH, N.S. – FIELD, C.B. – HUNGATE, B.A. – JOBBÁGY, E.G. – KUEPPERS, L.M. – NOSETTO, M.D. – PATAKI, D.E. (2008): Protecting climate with forests. Environ. Res. Lett. 3 044006 (5pp) doi: 10.1088/1748-9326/3/4/044006

JACOB, D. – KOTOVA, L. – LORENZ, P. – MOSELEY, C. – PFEIFER, S. (2008): Regional climate modeling activities in relation to the CLAVIER project. Időjárás 112: 141–153.

JACOB, D. – BÄRRING, L. – CHRISTENSEN, O.B. – CHRISTENSEN, J.H. – DE CASTRO, M. – DÉQUÉ, M. – GIORGI, F. – HAGEMANN, S. – HIRSCHI, M. – JONES, R. – KJELLSTRÖM, E. – LENDERINK, G. – ROCKEL, B. – SÁNCHEZ, E. – SCHÄR, C. – SENEVIRATNE, S.I. – SOMMOT, S. – VAN ULDEN, A. – VAN DEN HURK, B. (2007): An inter-comparison of regional climate models for Europe: model performance in present-day climate. Climatic Change 81: 31–52.

JACOB, D. – ANDRAE, U. – ELGERED, G. – FORTELIUS, C. – GRAHAM, L.P. – JACKSON, S.D. – KARSTENS, U. – KOEPKEN, CHR. – LINDAU, R. – PODZUN, R. – ROCKEL, B. – RUBEL, F. – SASS, H.B. – SMITH, R.N.D – VAN DEN HURK, B.J.J.M. – YANG, X. (2001): A Comprehensive Model Intercomparison Study Investigating the Water Budget during the BALTEX-PIDCAP Period. Meteorology and Atmospheric Physics 77 (1–4): 19–43.

JUNGCLAUS, J.H. – KEENLYSIDE, N. – BOTZET, M. – HAAK, H. – LUO, J-J. – LATIF, M. – MAROTZKE, J. – MIKOLAJEWICZ, U. – ROECKNER, E. (2006): Ocean circulation and tropical variability in the coupled model ECHAM5/MPI-OM. J. Climate 19: 3952–3972.

VAN DER LINDEN P, MITCHELL J.F.B. (Eds) (2009): ENSEMBLES: Climate Change and its Impacts: Summary of research and results from the ENSEMBLES project., Met Office Hadley Centre, FitzRoy Road, Exeter EX1 3PB, UK; 160 p.

MAHMOOD, R. – PIELKE, R.A. – SR.HUBBARD, K.G. – NIYOGI, D. – BONAN, G. – LAWRENCE. – BAKER, B. – MCNIDER, R. – MCALPINE, C. – ETTER, A. – ET AL. (2010): Impacts of land use land cover change on climate and future research priorities. Bull Am Meteorol Soc, 91:37–46.

MÁTYÁS, CS. (2009): Ecological perspectives of climate change in Europe's continental, drought-threatened Southeast. In: Groisman, P.Y. – Ivanov, S.V. (eds.): Regional aspects of climate-terrestrial-hydrologic interactions in non-boreal Eastern Europe. NATO Science Series, Springer Verl. 31–42.

MÁTYÁS, CS. – BERKI, I. – CZÚCZ, B. – GÁLOS, B. – MÓRICZ, N. – RASZTOVITS, E. (2010): Future of beech in Southeast Europe from the perspective of evolutionary ecology. Acta Silv. Lign. Hung. 6: 91–110.

MISHRA, A.K. – SINGH, V.P. (2010): A review of drought concepts. J of Hydrology 391:202–216.

NAKICENOVIC N ET AL. (2000): IPCC Special Report on Emission Scenarios. Cambridge: Cambridge University Press, p 599.

DE NOBLET-DUCOUDRÉ, N. AND COAUTHORS (2012): Determining Robust Impacts of Land-Use-Induced Land Cover Changes on Surface Climate over North America and Eurasia: Results from the First Set of LUCID Experiments. J. Climate, 25, 3261–3281.

PAL, J.S. – GIORGI, F. – BI, X. (2004): Consistency of recent European summer precipitation trends and extremes with future regional climate projections. Geophys. Res. Lett. 2004, 31 L13202 (4p). doi: 10.1029/2004GL019836

PIELKE, R.A. – AVISSAR, SR.R. – RAUPACH, M. – DOLMAN, A.J. – ZENG, X. – DENNING, A.S. (1998): Interactions between the atmosphere and terrestrial ecosystems: influence on weather and climate. Global Change Biology 4: 461–475.

PIELKE, SR. – ROGER A. –PITMAN, A. –NIYOGI, D. – MAHMOOD, R. – MCALPINE, C. – HOSSAIN, F. – KLEIN GOLDEWIJK, K. – UDAYSANKAR NAIR, BETTS, R. – FALL, S. – REICHSTEIN, M. – KABAT, P. – DE NOBLET, N. (2011): Land use/land cover changes and climate: modelling analysis and observational evidence. WIREs Clim Change 2: 828–850.

PITMAN, A.J. (2003): The evolution off, and revolution in, land surface schemes designed for climate models. Int. J. of Climatol. 23: 479–510.

PONGRATZ, J. – REICK, C.H. – RADDATZ, T. – CALDEIRA, K. – CLAUSSEN, M. (2011): Past land use decisions have increased mitigation potential of reforestation. Geophys Res Lett 38 L15701. doi:10.1029/2007GL031296

RASZTOVITS, E. – MÓRICZ, N. – BERKI, I. – PÖTZELSBERGER, E. – MÁTYÁS, CS. (2012): Evaluating the performance of stochastic distribution models for European beech at low-elevation xeric limits. Időjárás 116(3): 173–194.

RECHID, D. – JACOB, D. (2006): Influence of monthly varying vegetation on the simulated climate in Europe. Meteorol Z. 15: 99–116.

RECHID, D. – RADDATZ, T.J. – JACOB, D. (2008a): Parameterization of snow-free land surface albedo as a function of vegetation phenology based on MODIS data and applied in climate modelling. Theor. Appl. Climatol. 95: 245–255.

RECHID, D. – HAGEMANN, S. – JACOB, D. (2008b): Sensitivity of climate models to seasonal variability of snow-free land surface albedo. Theor. Appl. Climatol. 95: 197–221. doi: 10.1007/s00704-007-0371-8

ROECKNER, E. – BROKOPF, R. – ESCH, M. – GIORGETTA, M. – HAGEMANN, S. – KORNBLUEH, L. – MANZINI, E. – SCHLESE, U. – SCHULZWEIDA, U. (2006): Sensitivity of simulated climate to horizontal and vertical resolution in the ECHAM5 atmosphere model. J. Clim. 19: 3771–3791.

SÁNCHEZ E, GAERTNER MA, GALLARDO C, PADORNO E, ARRIBAS A, CASTRO M (2007): Impacts of a change in vegetation description on simulated European summer present-day and future climates. Clim Dyn 29: 319–332.

TEULING, A.L. et al. (2010): Contrasting response of European forest and grassland energy exchange to heatwaves. Nature Geoscience 3: 722–727.

VIDALE PL, LÜTHI D, WEGMANN R, SCHÄR C (2007): European summer climate variability in a heterogeneous multi-model ensemble. Climatic Change 81: 209–232.

WRAMNEBY, A. – SMITH, B. – SAMUELSSON, P. (2010): Hot spots of vegetation-climate feedbacks under future greenhouse forcing in Europe. J. Geophys. Res. 115: D21119 doi:10.1029/2010JD014307

# Influence of Regeneration Method on the Yield and Stem Quality of Black Locust (*Robinia pseudoacacia* L.) Stands: a Case Study

Károly RÉDEI[a*] – Imre CSIHA[a] – Zsolt KESERŰ[a] – János GÁL[b]

[a] Hungarian Forest Research Institute, Sárvár, Hungary
[b] Institute of Forest Resource Management and Rural Development, University of West Hungary, Sopron, Hungary

**Abstract** – Black locust (*Robinia pseudoacacia* L.) is one of the most important forest tree species in Hungary, covering approximately 23% of the forested land and providing 25% of the annual timber output of the country. One third of these black locust stands are high forests (planted with seedlings) and the remainder coppices. According to the forestry regulations black locust stands can be regenerated both by root suckers and with seedlings in Hungary. This study investigates the influence of different regeneration methods on wood production, stem quality and health. Properly managed regeneration from root suckers produced a higher yield than regeneration from seedlings at a harvest age of 35–37 years. The results show that regeneration of black locust stands from root suckers can be recommended on good and medium quality sites without a decrease in yield or stem quality.

***Robinia pseudoacacia* L. / regeneration methods / yield / health condition**

**Kivonat – Különböző felújítási módok hatása akácosok hozamára: esettanulmány.** Az akác (*Robinia pseudoacacia* L.) az egyik legfontosabb állományalkotó fafaj Magyarországon, az erdőterület mintegy 23%-át borítja és az ország éves faanyag-termelésének 25%-át adja. Ezen akácállományok egyharmada szálerdő (mageredetű), a maradék pedig sarj eredetű. Magyarországon a szakmai előírások értelmében az akácosok magról és gyökérsarjról újíthatók fel. A dolgozat különböző felújítási módoknak a fatermésre, a törzsminőségre, illetve a fák egészségi állapotára gyakorolt hatását vizsgálja. A számított adatok szerint a gyökérsarjról történő szakszerű és gondos felújítás a 35–37 éves véghasználati korban magasabb hozamot eredményezett, mint a magról történő felújítás. A vizsgálati eredmények alapján az akácosok gyökérsarjról történő felújítása elsősorban a jó és közepes termőhelyeken ajánlott a fatermés csökkenése nélkül.

**Fehér akác / felújítási módok / fatermés / egészségi állapot**

## 1 INTRODUTION

*Robinia pseudoacacia* was the first forest tree species brought from North America to Europe (to France) sometime after 1601. Its rapid spread all over the world may be attributed to its adaptability to a wide range of conditions, favorable breeding properties, frequent and abundant seed production, excellent coppicing, fast growth and high yield (Keresztesi 1988).

[*]Corresponding author: redei.karoly@t-online.hu, H-9600 SÁRVÁR, Várkerület 30/A.

Black locust was introduced in Hungary between 1710 and 1720. The first large black locust stands were established at the beginning of the 19th century on the Great Hungarian Plain stabilizing the wind-blown sandy soil. In Hungary black locust occupied 37.000 ha in 1885, 109.000 ha in 1911, 186.000 ha in 1938 and 420.000 ha in 2010. At present, it is the most widely planted tree species in Hungary, covering 23% of the country's total forest area. One-third of these stands are high forests and two-thirds are of coppice. In the 1960s, Hungary had more black locust forests than all the rest of Europe.

The average per hectare volume in all black locust forests is 125 $m^3$/ha, with an average volume of 190 $m^3$/ha at harvest at an average harvest age of 31. Black locust forests in Hungary have been established on a range of sites; however only sites with adequate moisture supply and well aerated and loose structured soil, rich in nutrients and humus can produce good quality timber. On medium and poor sites, black locust produces fuel wood, fodder, poles and mine props as well as honey while protecting the soil and the environment (Boring et al. 1984, Keresztesi 1988, Bongarten et al. 1992, Rédei 1996).

Black locust produces abundant seed crops of long-lived viable seeds which however need treatment before they can germinate. At the same time, roots produce vigorous suckers.

The most important black locust growing regions in Hungary are in the south and southwest Transdanubia (hill-ridges of Vas-Zala county, hill-ridges Somogy county), the plain between the Danube and Tisza rivers (Central Hungary) and north-east Hungary (Nyírség region) (*Figure 1*).

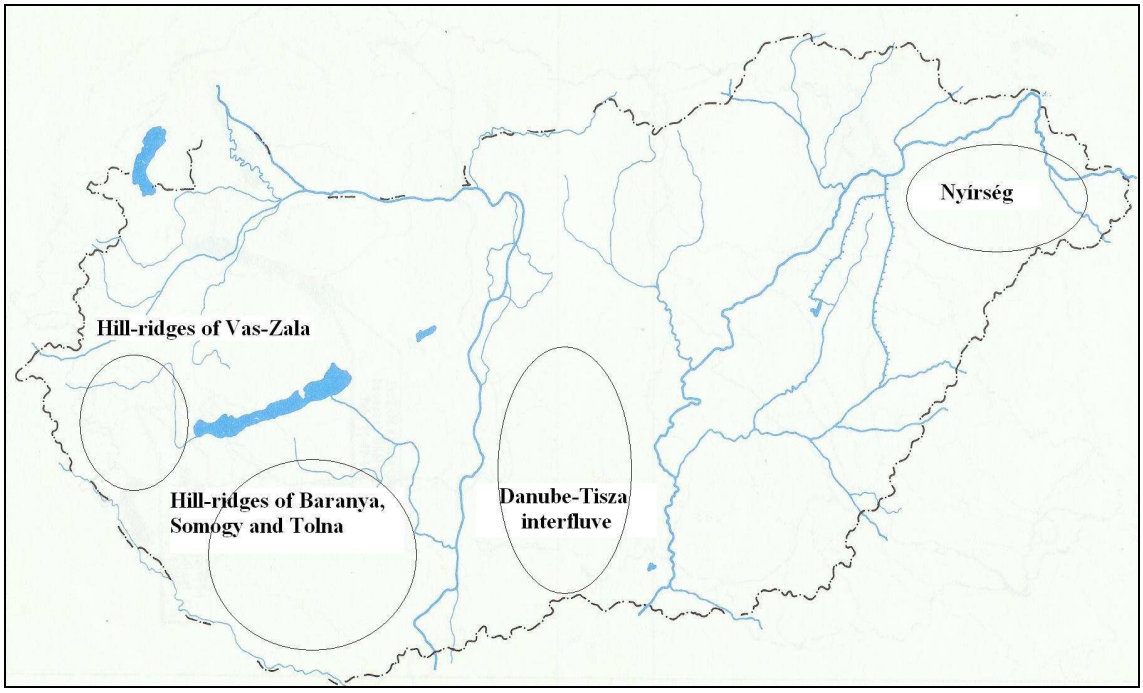



*Figure 1. The main growing regions of black locust (Robinia pseudoacacia* L.*) stands in Hungary*

In the future there are two regions in the world where black locust is likely to be grown in larger areas: the Mediterranean countries in Europe and China and Korea in Asia.

### 1.1 Stand establishment, tending and yield

Black locust requires well-drained soils with adequate moisture where the associated nitrogen-fixing *Rhizobium* bacteria are able to thrive. That is why soil preparation (total or partial) to improve aeration and the water regime of the soil and tilling between rows may

become necessary. It can grow on a wide range of soil types, except on very dry or very heavy soils. The aeration and water regime of the subsoil have a major influence on growth. In general, it prefers loose, structured soils, especially silty and sandy loams. Growth is very slow where the subsoil is compact and waterlogged, especially if the root zone is less than 35 cm; excessively dry sites are also very unfavorable (Eigel et al. 1980, Keresztesi 1988).

The most popular planting spacing for black locust in Hungary is 2.4 m by 0.7 to 1.0 m, requiring at least 4000 seedlings/ha. Black locust stands are often regenerated by coppice (from root suckers). In young stands of coppice origin, a cleaning operation should be done to adjust spacing when the stands are 3–6 years old and should reduce the stock below 5000 stems/ha.

Black locust is a fast-growing tree, and is able to quickly close canopy openings created by tending operations up to the age of 10–15 years; such closure is much slower in later years. Height growth peaks within the first five years, and diameter growth peaks in the first decade. The maximum current annual increment is near age 20; while mean annual increment culminates between age 35–37 years (based on the yield table Rédei – Gál 1985).

A growing space index is used to determine tending and thinning intensity. This index expresses the mean distance between trees (assuming triangular pattern) as a percentage of mean height after tending and thinning. The mean value of this index should be 23–24% for black locust stands. Crop trees should be pruned, so after selective thinning, stems must be free of branches up to a height of 4–6 m.

The objective of tending is to produce a high proportion of good quality saw-logs from stands on good sites; some saw-logs and a high proportion of poles and props from stands on medium quality sites and poles, props and other small-dimension industrial wood from the rest of the stands (Rédei 1992).

## 2 STUDY OBJECTIVE

Several experimental plots were established in Hungary in order to investigate the influence of different regeneration methods on wood production and quality in black locust stands (Keresztesi 1965, 1987, Rédei 1997). The purpose was to find the best regeneration method for these stands. Unfortunately most of the plots were established without replication; yet they still provide us with valuable and long term information about the growth and yield of black locust under different regeneration circumstances.

## 3 MATERIAL AND METHODS

### 3.1 The study area

In the area between the Danube and Tisza rivers (see *Figure 1*), three experimental plots were established in the spring of 1968, each 0.24 hectare. According to the Hungarian site classification, the main ecological characteristics of the study area are as follows:

- forest steppe climate zone (humidity is less than 50% in July at 2 pm, annual precipitation is less than 550 mm);
- hydrology: water loosing site (with lack of groundwater influence);
- genetic soil type: sand with humus content.

The latitude and longitude coordinates of the experimental site are *N 47.11, E 19.30.*

### 3.2 Treatments

Three different regeneration methods were used after harvest with stump extraction (Keresztesi 1987):

- Treatment I: Regeneration from suckers developed from thick roots (root diameter between 3 and 6 cm);
- Treatment II: Regeneration from suckers developed from thin roots (root diameter between 1 and 3 cm) with removal of thick roots;
- Treatment III: Regeneration with seedlings planted into deep-ploughed soil.

Harvest and tending operations have been done in accordance with intensive silviculture prescriptions of the time. Two cleanings were done at age of 5 and 10, a selective thinning at age of 15 and a late thinning at age 20. The trees on the plots were harvested at age of 34 and regenerated with the three different methods described above.

The yield class of the stand is II on a relative scale of I to VI, where I is the best (Rédei – Gál 1985).

### 3.3 Calculation of the stand parameters

DBH and height were measured for each tree on each plot at the age of 6, 17, 24, 29 and 34; then stem number, basal area, stand volume per hectare and average tree volume were calculated. Tree volume was calculated using the volume function based on the volume table for black locust (Sopp 1974):

$$V=\frac{d^2 \cdot h^{po+1}}{(h-1{,}3)^{po} \cdot 10^8} \cdot (p_1 \cdot d \cdot h + p_2 \cdot d + p_3 \cdot h + p_4) \quad (1)$$

where $d$ is d.b.h.(cm),
$h$ is tree height (m),
$p_o = 4$
$p_1 = -0.6326$
$p_2 = 20.23$
$p_3 = 0.00$
$p_4 = 3034$

### 3.4 Tree/stem quality classification

Four stem quality classes (depending on utilization) were identified in this study:

Class 1: Trees providing high quality industrial wood: straight, cylindrical, healthy stems, reaching to the top of the crown. Crooks are tolerated in one dimension only, not more than twice the stem diameter. The lower two-thirds of the tree must be suitable for saw logs and free of live branches.

Class 2: Trees providing lower quality industrial wood: the stem is straight, forks are tolerated, but only if they are in the uppermost third of the tree. Crooks are tolerated in one dimension only, not more than four times the stem diameter.

Class 3: Trees suitable for short logs of poor quality: the stem is crooked and leaning. Crooks may reach six times the stem diameter in one dimension and minor crookedness in a second dimension is tolerated. Only short logs of poor quality and firewood can be produced from these trees.

Class 4: Trees suitable for firewood only: very crooked in more than one dimension, low branching, forked trees with stem defects, broken crown or stem rot.

For the investigation of butt rot caused by the fungus (*Fomes fraxineus* COOKE), 30 sample trees were selected in each plot. Rot on cut surface and upwards in the trunk was measured by cutting the stem along the vertical axis.

The stand value index showing the stem quality of the stands (SVI) was determined base on the following formula:

$$\text{SVI} = \frac{x_1 n_1 + x_2 n_2 + x_3 n_3 + x_4 n_4}{n_1 + n_2 + n_3 + n_4} \tag{2}$$

where $x_1, x_2, x_3, x_4$ = tree quality classes,

$n_1, n_2, n_3, n_4$ = tree numbers belonging to the single tree quality classes

## 4 RESULTS

*Table 1* shows a considerable difference in initial stocking after regenerations. The best stocking was from regeneration from large roots. The lowest density was in treatment II, i.e. regeneration from fine roots. This difference also shows up in per hectare volume at harvest. There is virtually no difference in average heights and diameters between treatments I and II, i.e. regeneration from suckers, and this also applies to mean tree volumes. The lowest values for these parameters are in treatment III, i.e. regeneration with seedlings. There is no difference in terms of stand value index, meaning that the method of regeneration does not have any effect on the quality of the trees. In all treatments a tendency toward increasing SVI indices with age.

*Table 1. Stand structure and quality parameters of the experimental plots*

| Age | Stem number per ha | Mean height | Mean DBH | Basal area | Volume | Mean tree volume | Stand-value index (SVI) |
|---|---|---|---|---|---|---|---|
| Y ear | | m | cm | $m^2$/ha | $m^3$/ha | $dm^3$ | |
| Treatment I Regeneration from suckers developed from thick roots | | | | | | | |
| 6 | 5060 | 6.2 | 4.2 | 7.1 | 35.2 | 7.0 | – |
| 17 | 1283 | 16.8 | 13.7 | 19.0 | 165.8 | 129.2 | 1.7 |
| 24 | 607 | 21.6 | 19.0 | 17.2 | 182.6 | 300.8 | 1.8 |
| 29 | 601 | 23.1 | 21.5 | 21.8 | 248.5 | 413.5 | 1.9 |
| 34 | 601 | 24.2 | 23.2 | 23.6 | 297.1 | 494.3 | 1.9 |
| Treatment II Regeneration from suckers developed from thin roots | | | | | | | |
| 6 | 3872 | 6.6 | 4.9 | 7.2 | 37.2 | 9.6 | - |
| 17 | 896 | 17.4 | 15.4 | 16.7 | 153.3 | 171.1 | 1.6 |
| 24 | 395 | 22.6 | 19.8 | 12.1 | 135.7 | 343.5 | 1.7 |
| 29 | 395 | 23.3 | 22.4 | 15.6 | 178.0 | 450.6 | 1.8 |
| 34 | 395 | 23.9 | 23.8 | 15.7 | 200.0 | 506.3 | 1.8 |
| Treatment III Regeneration with seedlings | | | | | | | |
| 6 | 4004 | 6.2 | 4.2 | 5.5 | 27.6 | 6.9 | – |
| 17 | 1225 | 16.5 | 13.7 | 18.0 | 155.6 | 127.0 | 1.7 |
| 24 | 607 | 21.6 | 18.6 | 16.6 | 177.0 | 291.6 | 1.7 |
| 29 | 596 | 22.4 | 20.9 | 20.5 | 228.0 | 382.5 | 1.8 |
| 34 | 596 | 22.8 | 21.5 | 22.9 | 258.9 | 434.4 | 1.8 |

*Figure 2* shows similar tendencies of height growth curves of seed and coppice origin. At the age of harvest, the mean height of the stand regenerated from suckers (treatment I and II) developed from thick roots is just as large as that of the stand regenerated from suckers developed from thin roots. The height growth of the stand regenerated with seedlings (treatment III) is somewhat lower than that of the stand regenerated from suckers.

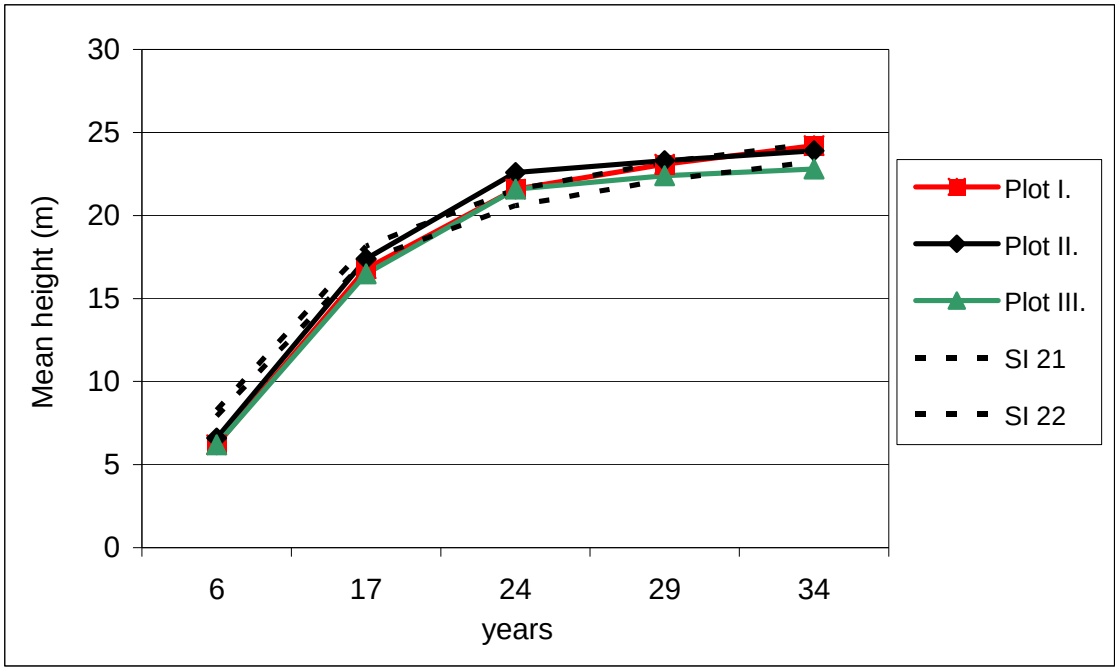



*Figure 2. Mean height values of stands along with site index curves (Plot I: regenerated from thick roots, Plot II regenerated from thin roots, Plot III: regenerated with seedlings, SI 21 and 22 are site index curves)*

To investigate possible differences in curve shapes and asymptotic values of height growth a growth function, the modified Chapman–Richards function was fitted to the height over age data. The function has the following form:

$$h = p_1(1 - e^{p_2 t})^{p_3} \tag{3}$$

where $h$ is the height
$t$ is the age
$p_1$, $p_2$ and $p_3$ are the parameters

Parameter $p_1$ is the asymptotic (maximum) value of the height growth; parameters $p_2$ and $p_3$ are responsible for the shape of the curve.

*Table 2. Parameter estimates for the height growth curves of the three different plots using the modified Chapman–Richards function*

| | Plot I | Plot II | Plot III |
|---|---|---|---|
| $p_1$ | 27.10 | 25.89 | 24.83 |
| $p_2$ | –0.07968 | –0.09681 | –0.09624 |
| $p_3$ | 1.5384 | 1.6927 | 1.7136 |

Parameter estimates in *Table 2* show that the height growth of plots II and III are similar in shape (nearly identical $p_2$ and $p_3$ parameters) at a slightly different asymptotic level, and the curve of plot I is different both in terms of asymptotic value and curve shape.

*Figure 2* also compares experimental plot height growth with site index curves (Rédei – Gál 1985) for site indices 21 and 22 (height at the age 25; dotted black lines in the graph). The growth pattern is very similar, so these long term observations also support the patterns of the site index curves.

*Figure 3* indicates that in spite of a difference in number of stems per ha between the plots regenerated from suckers sprouted from thick and thin roots there is no considerable difference in DBH of their crops. The DBH values of crops regenerated with seedlings are somewhat lower than that of crops regenerated from suckers but the deviation is negligible from a practical point of view.

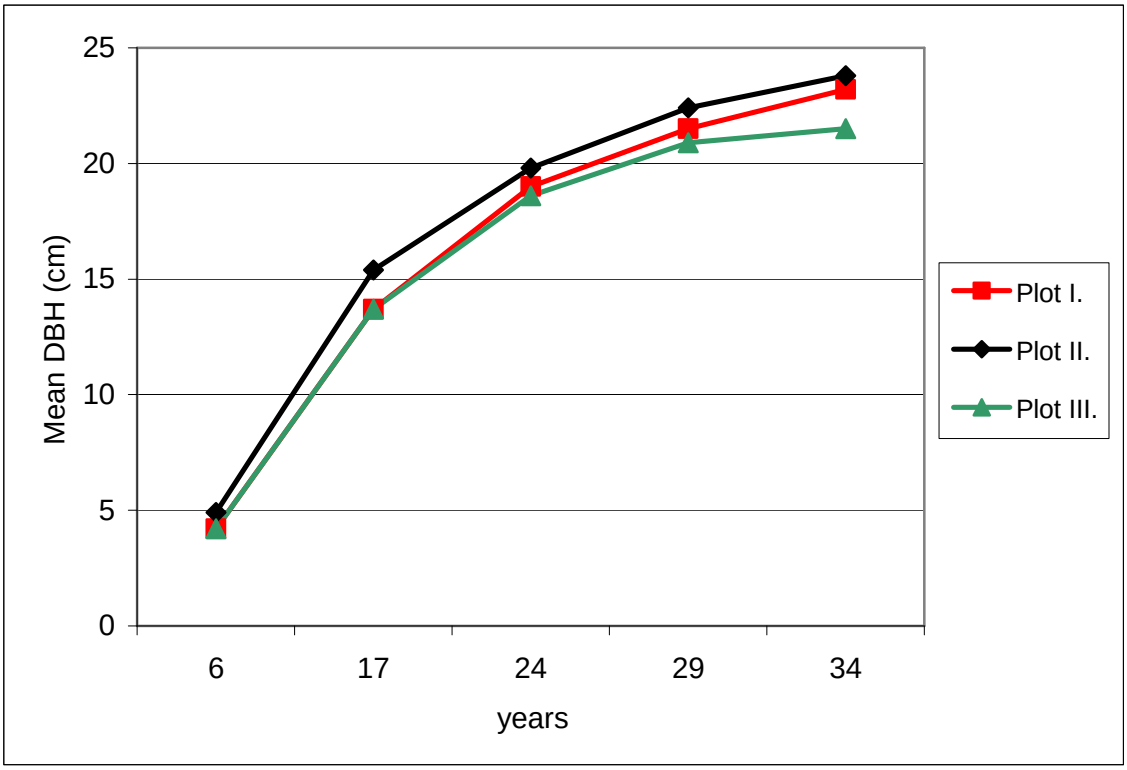



*Figure 3. DBH values of the stands*
*(Plot I: regenerated from thick roots, Plot II regenerated from thin roots, Plot III: regenerated with seedlings)*

The parameters of the Chapman–Richards function were also estimated for the DBH over age data. The parameters are in *Table 3*.

*Table 3. Parameter estimates for the DBH growth curves of the three different plots using the modified Chapman–Richards function*

| | Plot I | Plot II | Plot III |
|---|---|---|---|
| $p_1$ | 29.75 | 28.28 | 25.21 |
| $p_2$ | –0.05827 | –0.06772 | –0.07659 |
| $p_3$ | 1.6260 | 1.5995 | 1.8320 |

According to these data, stands raised from suckers produced higher average tree volumes in both cases than stands of seed origin.

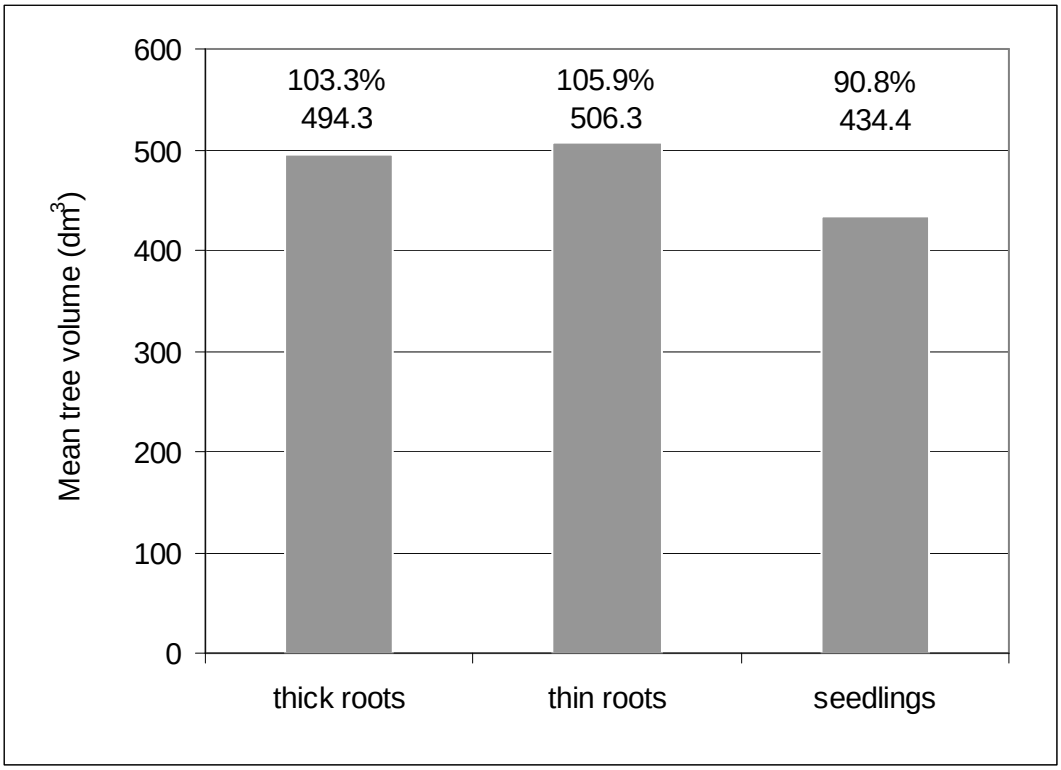



*Figure 4. Mean tree volume of the crops at age of 34 years in the treatments*

Investigations of stem quality of these experimental stands at harvest age 34 showed a stand value index 1.9 in treatment I (thick roots) and 1.8 in treatment II (thin roots) and III (seedlings). This suggests that differences in regeneration methods do not result in meaningful differences in the stand value index.

After harvesting, the degree of butt rot caused by *Fomes fraxineus* COOKE was investigated on 30 trees in each plot by determining the surface area of rot per cross section along the stem. 72% of the sample trees had rot extending up to 50 cm from the butt and 29% had rot between 50 and 100 cm up the stem. No trees had rot above 1 meter height. Overall 46% of the trees had butt rot on plot I, 43% on plot II, and 40% on plot III. This data would suggest that but rot is only slightly influenced by the regeneration method.

## 5 CONCLUSIONS

The results indicate that there is only a slight difference in stem quality and health status of stands regenerated by different methods. The differences that arose from treatments were mainly related to different initial stand densities.

Mean tree volumes of stands established by various coppicing methods and by planting have shown only minor differences at harvest. The stand of seed origin did not produce better stem quality than stands of coppice origin. The coppice stands did not produce less valuable assortments than stands of seed origin, therefore coppicing remains to be a viable management option under similar site conditions.

## REFERENCES

BONGARTEN, B.C. – HUBER D. A. – APSLEY, D.K. (1992): Environmental & genetic influences on short-rotation biomass production of black locust (*Robinia pseudoacacia* L.) in the Georgia Piedmont. Forest Ecology and Management 55:315–331.

BORING, L.R. – SWANK, W.T. (1984): The role of black locust (*Robinia pseudoacacia*) in forest succession. Journal of Ecology 72: 749766.

EIGEL, R. A. – WITTWER, R. F. – CARPENTER, S. B. (1980): Biomass and nutrient accumulation in young black locust stands established by direct seeding on surface mines in eastern Kentucky. In GARETT, H. E. – COX, G. S. (eds.): Proceedings, Central Hardwood Forestry Conference 111. University of Missouri, Columbia, 337–346.

HANOVER, J. (1992): Black Locust: A historical and future perspective. proceedings: International conference on black locust: Biology, culture and utilization. Michigan State University, MI. 7–18.

HUNTLEY, J. C. (1990): *Robinia pseudoacacia* L. black locust. In: BURNS, R. M. – HONKALA, B. H. (Tech. Coord.): Silvics of North America. Volume 2. Hardwoods. Agric. Handb. 654. Washington, DC: U.S. Department of Agriculture, Forest Service: 755–761.

KERESZTESI, B. (1965): Robinia growing in Hungary, Akadémiai Kiadó. Budapest, 665 p.

KERESZTESI, B. – BUJTÁS, Z. – BURJÁN, Á. – LESSÉNYI, B. – SZONTÁGH, P. (1987): Az Ófehértó 2 F akácfelújítási kísérlet értékelése és eredményei [The evaluation and results of the trial carried out on Robinia regeneration]. Erdészeti Kutatások 79. p. 7–17 (In Hungarian)

KERESZTESI, B. (1988): The black locust. Akadémiai Kiadó. Budapest.

RÉDEI, K. – GÁL, J. (1985): Akácosok fatermése [Yield of robinia stands]. Erdészeti Kutatások 77. p. 195–203. (In Hungarian)

RÉDEI, K. (1992): Management of black locust stands in Hungary. proceedings: International conference on black locust: Biology, culture and utilization. Michigan State University, MI. 38–43.

RÉDEI, K. (1996): Yield study relations of regeneration of Robinia (*Robinia pseudoacacia* L.) stands. In: SKOVSGAARD. J. P. – JOHANNSEN V. K. (eds.): Modelling regeneration success early growth of forest stands. Proceedings from the IUFRO Conference. Horsholm, 105–111.

RÉDEI, K. (1997): The effect of regeneration methods on the yield of black locust (*Robinia pseudoacacia* L.) stands in Hungary. Silva Lusitana 5(1):71–77. EFN. Lisboa. Portugal.

SOPP, L. (1974): Volume tables. Mezőgazdasági Kiadó. Budapest, 89–93.

VERSITA DOI: 10.2478/v10303-012-0009-0 Acta Silv. Lign. Hung., Vol. 8 (2012) 113–122

# Poroid Fungi of Hungary in the Collection of Zoltán Igmándy

Ilona SZABÓ*

Institute of Silviculture and Forest Protection, University of West-Hungary, Sopron, Hungary

**Abstract** – Zoltán Igmándy (1925–2000), prominent Hungarian mycologist, worked as professor of forest protection at the University of West-Hungary Sopron. His main research area was the investigation of wood-inhabiting poroid fungi of Hungary, their occurrence and importance in forest pathology and wood protection. During his 40 years of scientific activity Igmándy created a rich fungal collection (herbarium Z. Igmándy) which includes the complete polypore mycota known to occur in Hungary until 1990. The paper provides a brief compendium of the collection, a list of species characterised by the number of specimens, number of sampling locations and enumeration of the hosts and substrata of the specimens.

**forest pathology / mycology / wood decay**

**Kivonat** – **Magyarország csövestaplói Igmándy Zoltán gyűjteményében.** Igmándy Zoltán (1925–2000) neves magyar mikológus az erdővédelemtan professzora volt a soproni Nyugat-magyarországi Egyetemen (1950–1990). Fő kutatási területe a csövestaplók magyarországi előfordulása és növénykórtani, faanyagvédelmi jelentősége volt. 40 éves tudományos tevékenysége során gazdag taplógomba gyűjteményt hozott létre, amely a Magyarországon előforduló, 1990-ig ismert, fán élő poroid taplógomba fajokat a teljesség igényével tartalmazza. A tanulmány a gyűjtemény rövid összegzése, a hazai csövestapló fajok listája a gyűjteményi darabszámmal, a gyűjtési helyek számával és a gyűjteménydarabok gazdanövényeinek, aljzatainak felsorolásával jellemezve.

**erdészeti növénykórtan / mikológia / korhadás**

## 1 INTRODUCTION

Zoltán Igmándy (1925–2000), researcher and professor of forest protection at the University of West-Hungary, investigated the polypore fungi from the beginning of his scientific career (1950) until his retirement in 1990. His research activity initiated the modern investigation and detailed inventory of polypores in Hungary. He studied the former data and reports in the literature and revised the previously existing specimens and collections of poroid fungi from Hungary and the Carpathian-Basin (Igmándy 1956, 1957a, 1981, 1987). Igmándy collected and documented some thousand fungal specimens, often with the contribution of the university and forestry colleagues, from all forest areas and most of the localities in Hungary. During his scientific activity of 40 years Igmándy created a vast collection. Although he also

* szaboi@emk.nyme.hu; H-9400 SOPRON, Bajcsy-Zs. u. 4.

collected several samples of corticoid and steroid fungi, only the specimens of poroid fungi were processed and identified. The data of the collection were partially published by Igmándy (1956, 1957, 1965, 1968a, 1968b, 1970) and were synthesised in his academic dissertation (Igmándy 1981). The most common polypore species were described in detail for the larger public in his book issued in 1991 (Igmándy 1991). The occurrence and phytopathological significance of the polypores for each of forest tree species important in Hungary also were described (Igmándy 1982). The polypore collection is kept at the Institute of Silviculture and Forest Protection of the University of West-Hungary Sopron. Several specimens and duplicates of selected species are deposited in the herbarium of the Hungarian Natural History Museum Budapest.

## 2 MATERIAL STUDIED

Igmándy kept a record of specimens in register books. He documented the collected fungi and characterised them in detail: collecting time and location, forest type and age, host or substrate, condition of the host, etc., name of collectors, species name, and identifier. The samples were recorded in separate registers for polypores from Hungary, polypores from other regions of the Carpathian-Basin, and for non-poroid fungal specimens. We processed the complete database from the register of the poroid fungi collected in Hungary. At the same time we checked the specimens in the herbarium created by Igmándy. The data were summarised elaborating the checklist of species published here.

## 3 RESULTS

The register of poroid fungi collected in Hungary contains 3219 entities from which Igmándy determined 2621 specimens belonging to 148 species. The specimens were collected from 1950 until 1989, except for a few collected earlier by other mycologists and determined or revised by Igmándy. For example the samples of some rare species as *Fibroporia vaillantii*, *Flaviporus brownii*, *Polyporus rhizophilus* and *Rigidoporus moeszii* from former Hungarian collections revised by Igmándy were included in this collection in order to complete the species range of poroid fungi known to occur in Hungary (Igmándy 1981).

For this paper we prepared a summary of the database of poroid fungi collected in Hungary. Species are listed alphabetically and accompanied by the following data: current name according the MycoBank and Index Fungorum databases, other synonym names used by Igmándy in the collection, number of specimens, number of locations, host genera and other substrata.

### 1.1 The list of poroid fungi of Hungary in the collection of Z. Igmándy

***Abortiporus biennis*** (Bull.) Singer 1944
syn. *Heteroporus biennis* (Bull.) Lázaro Ibiza
20 specimens, 11 locations, on dead *Acer*, *Quercus, Robinia, Salix, Tilia*, and on soil.

***Abortiporus fractipes*** (Berk. et. M.A. Curtis) Bondartsev 1959
1 specimen, 1 location, on dead *Picea*.

***Albatrellus cristatus*** (Schaeff.) Kotl. et Pouzar 1957
syn. *Caloporus cristatus* (Schaeff.) Quél.
3 specimens, 1 location, on dead Picea, and on soil.

***Albatrellus pes-caprae*** (Pers.) Pouzar 1966
syn. *Caloporus pes-caprae* (Pers.) Pilát
1 specimen, 1 location, on soil.

***Antrodia albida*** (Fr.) Donk 1966
syn.: *Trametes albida* (Fr.) Bourdot et Galzin
syn. *Coriolellus albidus* (Fr.) Bondartsev
syn. *Coriolellus serpens* (Fr.) Bondartsev
6 specimens, 4 locations, on dead *Fagus, Quercus*.

***Antrodia gossypium*** (Speg.) Ryvarden 1973
syn. *Fibroporia gossypia* (Speg.) Parmasto
1 specimen, 1 location, on bult-in conifer wood.

***Antrodia lenis*** (P. Karst.) Ryvarden 1973
syn. *Poria lenis* (P. Karst.) Sacc.
2 specimens, 2 locations, on dead *Picea, Quercus*.

***Antrodia malicola*** (Berk. et M.A. Curtis) Donk 1966
syn. *Coriolellus malicola* (Berk. et M.A. Curtis) Murrill
5 specimens, 5 locations, on dead *Fagus, Populus, Quercus*.

***Antrodia ramentacea*** (Berk. et Broome) Donk 1966
syn. *Coriolellus ramentaceus* (Berk. et Broome) Domanski
12 specimens, 8 locations, on dead *Pinus*.

***Antrodia serialis*** (Fr.) Donk 1966
syn. *Trametes serialis* Fr.
syn. *Coriolellus serialis* (Fr.) Murrill
20 specimens, 7 locations, on dead *Abies, Picea, Pinus* and on built-in conifer wood.

***Antrodia sinuosa*** (Fr.) P. Karst. 1881
syn. *Coriolellus sinuosus* (Fr.) A.K. Sarkar,
syn. *Coriolellus vaporarius* (Pers.) Domanski
8 specimens, 3 locations, on built-in conifer wood.

***Antrodia xantha*** (Fr.) Ryvarden 1973
syn. *Amyloporia xantha* (Fr.) Bondartsev et Singer
4 specimens, 1 location, on dead *Larix, Picea, Prunus*.

***Antrodiella foliaceodentata*** (Nikol.) Gilb. et Ryvarden 1993
syn. *Trametes foliaceodentata* (Nikol.) Domanski
1 specimen, 1 location, on dead *Fagus*.

***Antrodiella hoehnelii*** (Bres.) Niemelä 1982
syn. *Trametes hoehnelii* (Bres.) Pilát
9 specimens, 3 locations, on dead *Fagus, Carpinus*.

***Aurantiporus fissilis*** (Berk. et M.A. Curtis) H. Jahn ex Ryvarden 1978
syn. *Leptoporus fissilis* (Berk. et M.A. Curtis) Pilát
syn. *Tyromyces fissilis* (Berk. et M.A. Curtis) Donk
15 specimens, 8 locations, on living and dead *Aesculus, Carpinus, Fagus, Malus, Pinus, Populus, Salix, Sorbus*.

***Bjerkandera adusta*** (Willd.) P. Karst. 1879
syn. *Gloeoporus adustus* (Willd.) Pilát
53 specimens, 28 locations, on dead *Alnus, Betula, Carpinus, Crataegus, Fagus, Picea, Pinus, Platanus, Populus, Prunus, Quercus, Tilia, Ulmus*.

***Bjerkandera fumosa*** (Pers.) P. Karst. 1879
syn. *Gloeoporus fumosus* (Pers.) Pilát
35 specimens, 27 locations, on dead and rarely living *Acer, Carpinus, Celtis, Fagus, Juglans, Populus, Quercus, Robinia, Salix, Ulmus*.

***Ceriporia excelsa*** (S. Lundell) Parmasto 1959
2 specimens, 2 locations, on dead *Quercus*.

***Ceriporia mellita*** (Bourdot) Bondartsev et Singer 1941
1 specimen, 1 location, on dead *Fagus*.

***Ceriporia viridans*** (Berk. et Broome) Donk 1933
8 specimens, 4 locations, on dead *Fagus, Populus*.

***Ceriporiopsis gilvescens*** (Bres.) Domanski 1963
3 specimens, 1 location, on dead Fagus.

***Ceriporiopsis mucida*** (Pers.) Gilb. et Ryvarden 1985
Syn. *Fibuloporia donkii* Domanski
3 specimens, 3 locations, on dead *Quercus*.

***Ceriporiopsis resinascens*** (Romell) Domanski 1963
2 specimens, 2 locations, on dead *Fagus, Fraxinus*.

***Cerrena unicolor*** (Bull.) Murrill 1903
syn. *Trametes unicolor* (Bull.) Pilát
41 specimens, 25 locations, on dead *Acer, Aesculus, Carpinus, Fagus, Juglans, Prunus, Quercus, Robinia, Tilia*.

***Climacocystis borealis*** (Fr.) Kotl. et Pouzar 1958
3 specimens, 3 locations, on dead *Pinus, Picea*.

***Coltricia perennis*** (L.) Murrill 1908
syn. *Polystictus perennis* (L.) Fr.
9 specimens, 6 locations, on soil.

***Coriolopsis gallica*** (Fr.) Ryvarden 1973
Syn. *Trametes gallica* Fr.
24 specimens, 18 locations, on dead *Acer, Fagus, Fraxinus, Populus, Prunus, Quercus, Sambucus, Ulmus*.

***Coriolopsis trogii*** (Berk.) Domanski 1968
syn. *Trametes gallica* Fr, f. *trogii* Berk.
syn. *Trametes trogii* Fr.
*51 specimens, 39 locations, on dead Betula, Liliodendron, Populus, Quercus, Salix.*

***Daedalea quercina*** (L.) Pers. 1821
syn. *Trametes quercina* (L.) Pilát
*20 specimens, 16 locations, on dead Castanea, Fraxinus, Quercus.*

***Daedaleopsis confragosa*** (Bolton) J. Schröt. 1888
syn. *Trametes confragosa* (Bolton) Jörstad
54 specimens, 28 locations, on dead *Alnus, Betula, Fagus, Fraxinus, Populus, Prunus, Quercus, Salix, Ulmus*.

***Daedaleopsis tricolor*** (Bull.) Bondartsev et Singer 1941
syn. *Trametes confragosa* (Bolton) Jörstad var. *Lenzites tricolor* Bull.
syn. *Daedaleopsis confragosa* var. *tricolor* (Bull.) Bondartsev et Singer
10 specimens, 10 locations, on dead *Betula, Prunus, Salix, Tilia*.

***Datronia mollis*** (Sommerf.) Donk 1966
syn. *Trametes mollis* (Sommerf.) Fr.
12 specimens, 9 locations, on dead *Acer, Fagus, Populus, Quercus, Salix*.

***Dichomitus campestris*** (Quél.) Domanski et Orlicz 1966
syn. *Trametes campestris* Quél.
2 specimens, 2 locations, on dead *Quercus*.

***Dichomitus squalens*** (P. Karst.) D.A. Reid. 1965
2 specimens, 2 locations, on dead *Pinus*.

***Diplomitoporus flavescens*** (Bres.) Domanski 1970
18 specimens, 11 locations, on dead Pinus.

***Donkioporia expansa*** (Desm.) Kotl. et Pouzar 1973
2 specimens, 2 locations, on built-in hard-wood.

***Fibroporia destructor*** (Schrad.) Parmasto 1968
1 specimen, 1 location, on built-in conifer wood.

***Fibroporia vaillantii*** (DC.) Parmasto 1968
1 specimen, 1 location, in building.

***Fistulina hepatica*** (Schaef.) With. 1821
9 specimens, 7 locations, on living and dead *Castanea, Quercus*.

***Flaviporus brownii*** (Humb.) Donk 1960
syn. *Leptoporus rufoflavus* (Berk. et M.A. Curtis) Pilát
syn. *Flaviporus rufoflavus* (Berk. et M.A. Curtis) Murrill
1 specimen, 1 location, on conifer mine-wood.

***Fomes fomentarius*** (L.) Fr. 1867
37 specimens, 23 locations, on living and dead *Aesculus, Betula, Carpinus, Fagus, Fraxinus, Populus, Prunus, Quercus, Salix, Ulmus*.

***Fomitopsis pinicola*** (Sw.) P. Karst. 1881
syn. *Fomes marginatus* (Fr.) Gillet
syn. *Fomitopsis marginata* (Pers. ex Fr.) P. Karst.
37 specimens, 25 locations, on dead *Abies*, Aesculus, Ailanthus, Alnus, Betula, Carpinus, *Fagus, Larix, Malus, Picea, Pinus, Populus, Prunus, Quercus, Tilia*.

***Fomitopsis rosea*** (Alb. et Schwein.) P. Karst. 1889
syn. *Fomes roseus* (Alb. et Schwein.) Cooke
8 specimens, 2 locations, on dead *Picea* and on built-in conifer wood.

***Ganoderma adspersum*** (Schulzer) Donk 1969
syn. *Ganoderma europaeum* Steyaert
syn. *Ganoderma linhartii* (Kalchbr.) Z. Igmándy
53 specimens, 40 locations, on living and dead *Abies, Acer, Aesculus, Betula, Carpinus, Celtis, Fagus, Fraxinus, Gleditsia, Gymnocladus, Morus, Picea, Platanus, Populus, Prunus, Quercus, Robinia, Salix, Tilia*.

***Ganoderma applanatum*** (Pers.) Pat. 1889
45 specimens, 37 locations, on dead *Acer (pseudoplatanus), Aesculus, Alnus, Betula, Carpinus, Fagus, Quercus, Picea, Populus, Prunus, Robinia, Salix, Tilia, Ulmus*.

***Ganoderma carnosum*** Pat. 1889
1 specimen, 1 location, on *Taxus*.

***Ganoderma cupreolaccatum*** (Kalchbr.) Z. Igmándy 1968
8 specimens, 7 locations, on living and dead *Acer, Fagus*.

***Ganoderma lucidum*** (Curtis) P. Karst. 1881
27 specimens, 19 locations, on living and dead *Acer, Carpinus, Platanus, Quercus, Salix, Taxus* and on soil.

***Ganoderma resinaceum*** Boud. 1889
syn. *Ganoderma lucidum* (Leys.) Karst. subsp. *resinaceum* (Boud.) Bourdot et Galzin
26 specimens, 11 locations, on living *Mahonia, Pyrus, Quercus, Robinia, Salix*.

***Gloeophyllum abietinum*** (Bull.) P. Karst. 1879
11 specimens, 3 locations, on dead *Picea* and on built-in conifer wood.

***Gloeophyllum odoratum*** (Wulfen) Imazeki 1943
syn. *Anisomyces odoratus* (Wulfen) Pilát
syn. *Osmoporus odoratus* (Wulfen) Singer
26 specimens, 14 locations, on dead *Picea, Pinus*.

***Gloeophyllum sepiarium*** (Wulfen) P. Karst. 1879
18 specimens, 9 locations, on *dead Larix, Picea, Pinus* and on built-in conifer wood.

***Gloeophyllum trabeum*** (Pers.) Murrill 1908
24 specimens, 14 locations, on dead *Acer, Betula, Fagus, Juniperus, Picea, Pinus, Populus, Quercus, Salix* and on built-in conifer and hardwood.

***Gloeoporus dichrous*** (Fr.) Bres. 1916
33 specimens, 25 locations, on dead *Ailanthus, Carpinus, Fagus, Populus, Pyrus, Quercus, Tilia*.

***Gloeoporus pannocinctus*** (Romell) J. Erikss. 1958
16 specimens, 7 locations, on dead *Ailanthus, Carpinus, Fagus, Populus*.

***Grifola frondosa*** (Dicks.) Gray 1821
5 specimens, 4 locations, on living *Quercus*.

***Hapalopilus croceus*** (Pers.) Bondartsev et Singer 1941
syn. *Phaeolus croceus* (Pers.) Pat.
6 specimens, 1 location, on living *Quercus*.

***Hapalopilus rutilans*** (Pers.) P. Karst. 1881
syn. *Phaeolus rutilans* (Pers.) Pat.
79 specimens, 56 locations, on dead *Acer, Betula, Carpinus, Fagus, Pinus, Prunus, Quercus, Salix, Tilia*.

***Heterobasidion annosum*** (Fr.) Bref. 1888
syn. *Fomes annosus* (Fr.) Cooke
80 specimens, 49 locations, on dead ***Abies***, *Alnus, Betula, Chaenomeles, Corylus, Cydonia, Larix, Picea, Pinus, Prunus, Pseudotsuga, Ribes, Tilia*.

***Inonotus cuticularis*** (Bull.) P. Karst. 1879
syn. *Xanthochrous cuticularis* (Bull.) Pat.
74 specimens, 44 locations, on living and dead *Acer, Aesculus, Fagus, Quercus, Morus, Populus*.

***Inonotus dryadeus*** (Pers.) Murrill 1908
syn. *Xanthochrous dryadeus* (Pers.) Z. Igmándy
17 specimens, 10 locations, on living and dead *Quercus*.

***Inonotus dryophilus*** (Berk.) Murrill 1904
syn. *Xanthochrous dryophylus* (Berk.) Z. Igmándy
11 specimens, 5 locations, on living *Quercus.*

***Inonotus hispidus*** (Bull.) P. Karst. 1889
syn. *Xanthochrous hispidus* (Bull.) Pat.
22 specimens, 15 locations, on living and dead *Aesculus, Fraxinus, Juglans, Malus, Morus, Platanus, Quercus*.

***Inonotus nidus-pici*** Pilát ex Pilát 1942
syn. *Xanthochrous nidus-pici* (Pilát) Z. Igmándy
31 specimens, 19 locations, on living *Acer, Aesculus, Fagus, Juglans, Quercus*.

***Inonotus nodulosus*** (Fr.) P. Karst. 1882
syn. *Xanthochrous radiatus* (Sow.) Pat. var. *nodulosus* (Fr.) Quél.
25 specimens, 14 locations, on dead *Fagus, Prunus, Quercus*.

***Inonotus obliquus*** (Ach. ex Pers.) Pilát 1942
syn. *Xanthochrous obliquus* (Pers.) B. et G.
26 specimens, 8 locations, on dead *Acer, Carpinus, Fagus, Quercus, Ulmus* and on living and dead *Betula* (f. asexualis).

***Inonotus radiatus*** (Sowerby) P. Karst. 1881
syn. *Xanthochrous radiatus* (Sow.) Pat.
29 specimens, 15 locations, on dead *Alnus, Carpinus, Fagus*.

***Inonotus rheades*** (Pers.) Bondartsev et Singer 1941
syn. *Xanthochrous rheades* (Pers.) Pat.
6 specimens, 3 locations, on dead *Populus tremula*.

***Irpex lacteus*** (Fr.) Fr. 1828
syn. *Trametes lactea* (Fr.) Pilát
15 specimens, 8 locations, on dead *Ailanthus, Fagus, Pinus, Populus, Prunus, Quercus, Robinia.*

***Ischnoderma benzoinum*** (Wahlenb.) P. Karst. 1881
8 specimens, 6 locations, on dead *Pinus, Picea*.

***Ischnoderma resinosum*** (Schrad.) P. Karst. 1879
14 specimens, 7 locations, on dead *Fagus, Quercus*.

***Junghuhnia lacera*** (P. Karst.) Niemelä et Kinnunen 2001
syn. *Chaetoporus radulus* (Pers. ex Fr.) Bondartsev et Singer
syn. *Junghuhnia separabillima* (Pouzar) Ryvarden
2 specimens, 2 locations, on dead *Populus*.

***Junghuhnia nitida*** (Pers.) Ryvarden 1972
syn. *Chaetoporus euporus* (P. Karst.) Bondartsev et Singer
3 specimens, 3 locations, on dead *Salix, Populus, Quercus*.

***Laetiporus sulphureus*** (Bull.) Murrill 1920
syn. *Grifola sulphurea* (Bull.) Pilát
19 specimens, 15 locations, on living and dead *Castanea, Populus, Prunus, Quercus, Robinia, Salix*.

***Lenzites betulina*** (L.) Fr. 1838
syn. *Trametes betulina* (L.) Pilát
33 specimens, 18 locations, on dead *Alnus, Betula, Carpinus, Cornus, Fagus, Picea, Populus, Quercus*.

***Lenzites warnieri*** Durieu et Mont. 1860
syn. *Lenzites reichardtii* Schulzer
7 specimens, 5 locations, on *dead Fraxinus, Quercus, Populus, Ulmus*.

***Meripilus giganteus*** (Pers.) P. Karst. 1882
syn. *Grifola gigantea* (Pers.) Pilát
31 specimens, 20 locations, on living and dead *Fagus, Quercus, Robinia*.

***Meruliopsis taxicola*** (Pers.) Bondartsev 1959
syn. *Poria taxicola* (Pers.) Bres.
syn. *Caloporus taxicola* (Pers.) Ryvarden
17 specimens, 12 locations, on dead *Pinus*.

***Onnia tomentosa*** (Fr.) P. Karst. 1889
1 specimen, 1 location, on soil.

***Osteina obducta*** (Berk.) Donk 1966.
syn. *Grifola ossea* (Kalchbr.) Pilát
1 specimen, 1 location, on dead *Larix*.

***Oxyporus latemarginatus*** (Durieu et Mont.) Donk 1966
syn. *Chaetoporus ambiguus* (Bres.) Bondartsev et Singer
20 specimens, 15 locations, on dead *Carpinus, Fagus, Fraxinus, Populus, Quercus, Robinia, Salix, Ulmus* and on living *Robinia*.

***Oxyporus populinus*** (Schumach.) Donk 1933
11 specimens, 5 locations, on dead *Acer, Fagus, Fraxinus, Sambucus*.

***Oxyporus ravidus*** (Fr.) Bondartsev et Singer 1941
1 specimen, 1 location, on dead *Picea*.

***Pachykytospora tuberculosa*** (Fr.) Kotl. et Pouzar 1963.
14 specimens, 5 locations, on dead *Quercus, Ulmus*.

***Perenniporia fraxinea*** (Bull.) Ryvarden 1978
syn. *Fomes fraxineus* (Bull.) Cooke
syn. *Haploporus cytisinus* (Berk.) Domanski
syn. *Perenniporia cytisina* (Berk.) n. c.
56 specimens, 43 locations, on living and dead *Platanus, Populus, Quercus, Robinia*.

***Perenniporia medulla-panis*** (Jack.) Donk 1967
19 specimens, 12 locations, on built-in hardwood (*Quercus, Robinia*).

***Phaeolus schweinitzii*** (Fr.) Pat. 1900
23 specimens, 9 locations, on living and dead *Larix, Picea, Pinus, Prunus* and on soil.

***Phellinus conchatus*** (Pers.) Quél. 1886
17 specimens, 12 locations, on dead *Acer, Fagus, Picea?, Prunus, Salix*.

***Phellinus contiguus*** (Pers.) Pat. 1928.
93 specimens, 48 locations, on dead *Ailanthus, Carpinus, Fagus, Fraxinus, Picea, Populus, Prunus, Quercus, Robinia* and on built-in wood.

***Phellinus ferruginosus*** (Schrad.) Pat. 1900
9 specimens, 7 locations, on dead *Carpinus, Fagus, Quercus, Robinia*.

***Phellinus hartigii*** (Allesch. et Schnabl) Pat. 1903
syn. *Phellinus robustus* f. *hartigii* (Allesch. et Schnabl) Bourdot et Galzin
14 specimens, 3 locations, on dead *Abies, Picea, Thuja*.

***Phellinus igniarius*** (L.) Quél. 1886
30 specimens, 22 locations, on living and dead *Alnus, Populus, Robinia, Salix*.

***Phellinus laevigatus*** (Fr.) Bourdot et Galzin 1928
1 specimen, 1 location, on dead *Betula*.

***Phellinus pilátii*** Cerny 1968
5 specimens, 2 locations, on living and dead *Populus alba*.

***Phellinus punctatus*** (P. Karst.) Pilát 1942
21 specimens, 12 locations, on living and dead *Carpinus, Corylus, Gleditsia, Juniperus, Lonicera, Morus, Populus, Salix, Syringa*.

***Phellinus rhamni*** (Bondartseva) H. Jahn 1967
2 specimens, 2 locations, on dead *Rhamnus*.

***Phellinus ribis*** (Schumach.) Quél. 1886
24 specimens, 11 locations, on living *Cornus, Crataegus, Euonymus, Ligustrum, Ribes*.

***Phellinus robustus*** (P. Karst.) Bourdot et Galzin 1928.
40 specimens, 25 locations, on living and dead *Amorpha, Castanea, Carpinus, Quercus, Robinia, Syringa, Viburnum*.

***Phellinus torulosus*** (Pers.) Bourdot et Galzin 1925
51 specimens, 28 locations, on living and dead *Acer, Castanea, Cornus, Crataegus, Euonymus, Fraxinus, Larix, Picea, Prunus, Pyrus, Quercus, Robinia, Salix*.

***Phellinus tremulae*** (Bondartsev) Bondartsev et P.N. Borisov 1953
syn. *Phellinus igniarius* (L. ex Fr.) f. *tremulae* Bondartsev
18 specimens, 12 locations, on living and dead *Populus*.

***Phellinus tuberculosus*** (Baumg.) Niemelä 1982
syn. *Phellinus igniarius* subsp. *pomaceus* (Pers.) Quél.
syn. *Phellinus pomaceus* (Pers.) Maire 1933
20 specimens, 16 locations, on living and dead *Prunus*.

***Phellinus viticola*** (Schwein.) Donk 1966
syn. *Phellinus isabellinus* (Fr.) Bourdot et Galzin
2 specimens, 2 locations, on dead *Pinus*.

***Piptoporus betulinus*** (Bull.) P. Karst. 1881
syn. *Placodes betulinus* (Bull.) Quél.
14 specimens, 12 locations, on living and dead *Betula*.

***Piptoporus quercinus*** (Schrad.) P. Karst. 1881
syn. *Placodes quercinus* (Schrad.) Quél.
4 specimens, 2 locations, on living *Quercus*.

***Polyporus alveolaris*** (DC.) Bondartsev et Singer 1941
syn. *Polyporus mori* (Pollini) ex Fr.
9 specimens, 8 locations, on dead *Cornus, Crataegus, Fagus, Maclura, Populus, Robinia, Syringa*.

***Polyporus arcularius*** (Batsch) Fr. 1821
41 specimens, 30 locations, on dead *Aesculus, Alnus, Carpinus, Fagus, Populus, Prunus, Quercus, Robinia, Salix*.

***Polyporus badius*** (Pers.) Schwein. 1832
syn. *Polyporellus picipes* (Fr.) Karst.
15 specimens, 8 locations, on dead *Acer, Fagus, Populus, Quercus, Salix*.

***Polyporus brumalis*** (Pers.) Fr. 1818
9 specimens, 7 locations, on dead *Alnus, Betula, Carpinus, Prunus, Quercus*.

***Polyporus ciliatus*** Fr. 1815.
2 specimens, 2 locations, on dead *Fagus*.

***Polyporus melanopus*** (Pers.) Fr. 1821
2 specimens, 2 locations, on dead *Fagus*, on soil.

***Polyporus rhizophilus*** Pat. 1894
3 specimens, 3 locations, on soil.

***Polyporus squamosus*** (Huds.) Fr. 1821
syn. *Polyporellusus squamosus* (Huds.) Karst.
15 specimens, 9 locations, on living and dead *Acer, Aesculus, Fagus, Juglans, Populus, Quercus, Sambucus, Ulmus*.

***Polyporus tuberaster*** (Jack. ex Pers.) Fr. 1815
3 specimens, 2 locations, on dead *Carpinus, Fagus*.

***Polyporus umbellatus*** (Pers.) Fr. 1821
syn. *Grifola umbellata* (Pers.) Pilát
5 specimens, 3 locations, on dead *Betula, Quercus*.

***Polyporus varius*** (Pers.) Fr. 1821
syn. *Polyporellus elegans* (Bull.) Karst.
11 specimens, 5 locations, on dead *Fagus*.

***Porodaedalea pini*** (Brot.) Murrill 1905
syn. *Phellinus pini* (Thore) Pilát
9 specimens, 4 locations, on living *Pinus*.

***Postia balsamea*** (Peck) Jülich 1982
syn. *Leptoporus kymatodes* (Rostr.) Pilát
syn. *Tyromyces balsameus* (Peck) Murrill
9 specimens, 2 locations, on dead *Juniperus, Picea, Pinus*.

***Postia caesia*** (Schrad.) P. Karst. 1881
syn. *Leptoporus caesius* (Schrad.) Quél.
syn. *Tyromyces caesius* (Schrad.) Murrill
16 specimens, 6 locations, on dead *Abies, Fagus, Picea, Pinus, Quercus, Tilia*.

***Postia fragilis*** (Fr.) Jülich 1982
syn. *Tyromyces fragilis* (Fr.) Donk
4 specimens, 4 locations, on dead *Picea, Pinus.*

***Postia placenta*** (Fr.) M.J. Larsen et Lombard 1986
syn. *Ceriporia placenta* (Fr.) Domanski
1 specimen, 1 location, on dead *Pinus*, on soil.

***Postia simanii*** (Pilát ex Pilát) Jülich 1982
syn. *Tyromyces simanii* (Pilát) Parmasto
1 specimen, 1 location, on dead *Populus*.

***Postia stiptica*** (Pers.) Jülich 1982
syn. *Leptoporus stipticus* (Pers.) Quél.
syn. *Tyromyces stipticus* (Pers. ex Fr.) Kotl. et Pouzar
30 specimens, 12 locations, on dead *Abies, Alnus, Betula, Carpinus, Fagus, Picea, Pinus.*

***Postia subcaesia*** (A. David) Jülich 1982
syn. *Tyromyces subcesius* (David)
4 specimens, 2 locations, on dead *Carpinus, Populus, Salix*.

***Pycnoporus cinnabarinus*** (Jack.) P. Karst. 1881
syn. *Trametes cinnabarina* (Jack.) Fr.
18 specimens, 8 locations, on dead *Fagus, Prunus, Quercus*.

***Rigidoporus moeszii*** (Pilát ex Pilát) Pouzar 1966
syn. *Leptoporus moeszii* Pilát
2 specimens, 1 location, on dead *Robinia*.

***Rigidoporus sanguinolentus*** (Alb. et Schwein.) Donk 1966
4 specimens, 1 location, on dead *Abies, Picea*.

***Rigidoporus vitreus*** (Pers.) Donk 1966
syn. Rigidoporus vitreus (Pers. ex Fr.) Donk
1 specimen, 1 location, in cellar.

***Schizopora carneolutea*** (Rodway et Cleland) Kotl. et Pouzar 1979
syn. *Schizopora phellinoides* (Pilát) Domanski
56 specimens, 30 locations, on dead *Acer, Alnus, Carpinus, Fagus, Populus, Quercus, Salix, Ulmus*.

***Schizopora paradoxa*** (Schrad.) Donk 1967
syn. *Xylodon versiporus* (Pers.) Bondartsev
syn. *Poria versipora* (Pers.) Baxter
92 specimens, 38 locations, on dead *Abies, Acer, Betula, Carpinus, Fagus, Picea, Pinus, Populus, Prunus, Quercus, Robinia, Salix, Tilia*.

***Skeletocutis alutacea*** (J. Lowe) Jean Keller 1979
syn. *Fibuloporia alutacea* (Lowe) Christ.
1 specimen, 1 location, on dead *Picea*.

***Skeletocutis amorpha*** (Fr.) Kotl. et Pouzar 1958
syn. *Gloeoporus amorphus* (Fr.) Killerm.
22 specimens, 13 locations, on dead *Larix, Picea, Pinus*.

***Skeletocutis nivea*** (Jungh.) Jean Keller 1979
syn. *Leptoporus semipileatus* (Peck) Pilát
syn. *Incrustoporia nivea* (Jungh.) Ryvarden
7 specimens, 4 locations, on dead *Acer, Carpinus, Quercus, Salix*.

***Skeletocutis tschulymica*** (Pilát) Jean Keller 1979
syn. *Incrustoporia tschulymica* (Pilát) Domanski
5 specimens, 1 locations, on dead *Populus*.

***Spongipellis delectans*** (Peck) Murrill 1907
2 specimens, 1 location, on dead *Fagus*.

***Spongipellis litschaueri*** Lohwag 1931
syn. *Leptoporus irpex* (Schulzer) n. c.
syn. *Spongipellis irpex* (Schulzer) n. c.
39 specimens, 26 locations, on living and dead *Acer, Ailanthus, Populus, Quercus*.

***Spongipellis pachyodon*** (Pers.) Kotl. et Pouzar 1965
syn. *Trametes pachyodon* (Pers.) Pilát
1 specimen, 1 location, on living *Platanus*.

***Spongipellis spumeus*** (Sowerby) Pat. 1900
syn. *Leptoporus spumeus* (Sowerby) Pilát
23 specimens, 10 locations, on living and dead *Acer, Aesculus, Juglans, Quercus, Populus, Ulmus*.

***Trametes gibbosa*** (Pers.) Fr. 1838
25 specimens, 15 locations, on dead *Abies, Aesculus, Alnus, Carpinus, Fagus, Picea, Platanus, Populus, Quercus, Salix, Tilia*.

***Trametes hirsuta*** (Wulfen) Pilát 1939
35 specimens, 23 locations, on dead *Aesculus, Betula, Carpinus, Fagus, Fraxinus, Malus, Pinus, Populus, Prunus, Quercus, Ulmus*.

***Trametes ochracea*** (Pers.) Gilb. et Ryvarden 1987
syn. *Trametes zonata* (Nees. ex Fr.) Pilát
4 specimens, 3 locations, on dead *Betula, Populus*.

***Trametes pubescens*** (Schumach.) Pilát 1939
14 specimens, 9 locations, on dead *Betula, Fagus, Populus, Prunus, Salix*.

***Trametes suaveolens*** (L.) Fr. 1838
32 specimens, 22 locations, on living and dead *Populus, Salix*.

***Trametes versicolor*** (L.) Lloyd 1921
29 specimens, 15 locations, on dead *Alnus, Betula, Carpinus, Fagus, Fraxinus, Picea, Pinus, Populus, Prunus, Quercus, Salix, Syringa*.

***Trametopsis cervina*** (Schwein.) Tomsowski 2008
syn. *Trametes cervina* (Schwein.) Bres.
25 specimens, 11 locations, on dead *Fagus, Quercus*.

***Trichaptum abietinum*** (Dicks.) Ryvarden 1972
syn. *Trametes abietina* (Dicks.) Pilát
syn. *Hirschioporus abietinus* (Dicks.) Donk
16 specimens, 7 locations, on dead *Abies, Picea, Pinus*.

***Trichaptum biforme*** (Fr.) Ryvarden 1972
syn. *Trametes biformis* (Fr.) Pilát
syn. *Hirschioporus pergamenus* (Fr.) Bondartsev et Singer
24 specimens, 12 locations, on dead *Fagus, Quercus, Tilia*.

***Trichaptum fuscoviolaceum*** (Ehrenb.) Ryvarden 1972
syn. *Trametes abietina* var. *fusco-violacea* Ehrenb.
syn. *Hirschioporus fusco-violaceus* (Ehrenb. ex Fr.) Donk
24 specimens, 18 locations, on dead *Picea, Pinus*.

***Tyromyces chioneus*** (Fr.) P. Karst. 1881
2 specimens, 2 locations, on dead *Fagus, Quercus*.

***Tyromyces lacteus*** (Fr.) Murrill 1907
syn. *Leptoporus lacteus* (Fr.) Quél.
16 specimens, 11 locations, on dead *Carpinus, Fagus, Malus, Populus, Quercus, Salix*.

**Acknowledgements:** The author thanks R. Dankóné and J. Kovács for their invaluable help with the processing of the data and checking the specimens in the collection.

## REFERENCES

IGMÁNDY, Z. (1956): Adatok a Kárpát-medence *Polyporaceae* flórájához I–II. [Data to the *Polyporaceae* of the Carpathian-Basin I–II.] Botanikai Közlemények 46: 306–306. (in Hungarian).

IGMÁNDY, Z. (1957): Adatok a Kárpát-medence *Polyporaceae* flórájához III. [Data to the *Polyporaceae* of the Carpathian-Basin III.] Botanikai Közlemények 47: 101–103. (in Hungarian).

IGMÁNDY, Z. (1965): Magyarország taplógombái I. (Polypori Hungariae I.) Erdészeti és Faipari Egyetem Tudományos Közleményei 1965 (1–2): 201–221. (in Hungarian).

IGMÁNDY, Z. (1968a): Die Porlinge Ungarns und ihre phytopathologische Bedeutung II. (Polipori Hungariae II.) Acta Phytopath. Acad. Sci.Hung. 3 (2): 221–239.

IGMÁNDY, Z. (1968b): Die Porlinge Ungarns und ihre phytopathologische Bedeutung III. (Polipori Hungariae III.) Acta Phytopath. Acad. Sci.Hung. 3 (3): 349–359.

IGMÁNDY, Z. (1970): Die Porlinge Ungarns und ihre phytopathologische Bedeutung IV. (Polipori Hungariae IV.). Acta Phytopath. Acad. Sci. Hung. 5 (2–4): 279–301.

IGMÁNDY, Z. (1981): Hazánk csövestapló (*Polyporaceae* s.l.) flórája és a fajok növénykórtani jelentősége. [The *Polyporaceae* s. l. of Hungary and their phytopathological significance.] MTA Dissertation, Budapest. 159 p. (in Hungarian).

IGMÁNDY Z. (1982): Fafajaink csövestaplói (*Polyporaceae* s.l.). [The *Polyporaceae* s. l. of our tree species.] Erdészeti és Faipari Egyetem Tudományos Közleményei 1982 (1): 5–24. (in Hungarian)

IGMÁNDY, Z. (1987): Schulzer hazánkban gyűjtött csövestaplóinak (*Polyporaceae* s.l.) revíziója. [The revision of the poroid fungi collected by Schulzer in Hungary.] Mikológiai Közlemények 1987 (2–3): 102–108. (in Hungarian).

IGMÁNDY, Z. (1991): A magyar erdők taplógombái. [The poroid fungi of the Hungarian forests.] Akadémiai Kiadó, Budapest. 112 p. (in Hungarian).

# Storage Problems of Poplar Chips from Short Rotation Plantations with Special Emphasis on Fungal Development

Zsuzsanna HORVÁTH[a] – Béla MAROSVÖLGYI[a] –
Christine IDLER[b] – Ralf PECENKA[b*] – Hannes LENZ[b]

[a] University of West Hungary, Sopron, Hungary
[b] Leibniz Institut für Agrartechnik, Potsdam-Bornim, Germany

**Abstract –** There are several problems in **s**toring wood chips freshly harvested from short rotation plantations, which result in quality losses as well as in dry matter and energy losses. The factors influencing the degradation of raw material are examined in this paper with special focus on fungal development. An excessive growth of fungi is connected to dry matter losses and also to an increased health risk during raw material handling.

The following factors were measured during 6 months storage of poplar wood chips depending on particle size: box temperature, moisture content, pH-value, appearance of fungi in the storage and the concentration of fungal particles in the air. The results show a close connection between particle size, temperature and attack of fungi. During the storage mesophilic and termophilic species of the genera *Alternaria, Aspergillus, Cladosporium, Mucor* and *Penicillium* appeared. The concentration of fungal particles is the highest for fine chips and decreases in bigger particles. There was a special focus on the investigation of the properties of coarse chips (G 50), which represent a good compromise between handling, storage losses and health risk due to fungal development.

**wood chips / poplar / storage / temperature / fungi**

**Kivonat – Rövid vágásfordulójú nyár apríték tárolási problémái, különös tekintettel a kifejlődő gombákra.** Frissen kitermelt rövid vágásfordulójú fafajokból előállított apríték tárolása során számos probléma merül fel, melyek eredménye minőség-, szárazanyag- és energiaveszteség. Ebben a tanulmányban a nyersanyag-degradációt befolyásoló tényezők kerültek vizsgálatra, különös tekintettel a kifejlődő gombákra. Ezek rendkívüli elszaporodásának köszönhető a szárazanyag-veszteség mellett a nyersanyag kezelése során fellépő, megnövekedett egészségügyi kockázat is.

Nyár faapríték hat hónapos tárolása folyamán az aprítékméret függvényében a következő tényezők mérése zajlott: a tárolón belüli hőmérséklet, nedvességtartalom, pH, a tárolóban megjelenő gombák és azok száma a levegőben. Az eredmények szoros összefüggést mutatnak, különösen az aprítékméret, a hőmérséklet és a gombák száma közt. A tárolás során mezofil és termofil gombák fejlődtek. A következő nemzetségekhez tartozó fajok voltak megfigyelhetők: *Alternaria, Aspergillus, Cladosporium, Mucor, Penicillium.* A gombák száma a finom aprítéknál volt a legmagasabb, ami az aprítékméret növekedésével csökkent. A kutatás folyamán különös szerepet kapott a középfinom apríték tulajdonságainak vizsgálata (G 50), mely kompromisszumos megoldást jelent a kezelés, tárolási veszteségek és a kifejlődő gombáknak köszönhető egészségügyi kockázatok között.

**faapríték / nyár / tárolás / hőmérséklet / gomba**

[*] Corresponding author: RPecenka@atb-potsdam.de, Max-Eyth-Allee 100, D-14469 POTSDAM

## 1 INTRODUCTION

The demand for wood chips from short rotation crops (SRC) such as poplar and willow for bioenergy has increased in recent years due to annually increasing total energy consumption and high prices for fossil resources. SRC chips have several advantages such as harvesting in winter, low fertilisation and maintenance needs, but some disadvantages as well (Jirjis 1995, Scholz et al. 2004, 2009). Storage at high moisture content influences the quality and storage losses of wood. Several factors have effects on storage processes. Many studies have shown that the biggest influencing factors are particle size, height of pile, air permeability and rain protection (Jirjis 2005, Scholz et al. 2005, Scholz – Idler 2005). Temperature development and moisture content in the storage pile, dry matter loss, quantity of fungal particles[1] in the air vary depending on the layout of the storage (Scholz – Idler 2001, Idler et al. 2004). The aim of this work was to determine the influence of the particle size on storage of poplar chips, to identify optimal forms of raw material preparation and storage design for minimum energy loss and health risks for operators. We also wanted to find an optimal length of chips, where the quality and mass losses are not as high as in common fine chips (G 30), but which are still easy usable for medium sized heating systems. Earlier studies have shown that the minimum energy loss and the lowest health risks for operators were reached by storing wood as chunks instead of chips (Scholz-Idler 2005, 2007). But chunks have to be comminuted before they can be burned and they are difficult to handle. Therefore, this study was done with special focus on coarse chips, which are still easy to handle and probably ensure lower mass losses during storage and reduced health risks than fine chips.

## 2 MATERIALS AND METHODS

The raw material used for this investigation was harvested from an 18 year old SRC plantation of poplar (Japan 105, 2 year rotation cycle) at the Leibniz Institute for Agricultural Engineering Potsdam-Bornim (ATB, Germany) in January 2012. The harvested material was chopped to different lengths immediately after they were felled and before storage. Three different particle sizes were produced and classified according to ÖNORM M 7133 for storage tests: fine chips G 30 (tractor mounted disk chipper FARMI CH150), coarse chips G 50 (tractor mounted ATB mower-chipper), and chunks G 100 (Chunk chopper DIEMER). The chips and chunks where stored immediately after harvest and prepared in separate rectangular boxes 2.5 m high and 2.0 m wide ($V$=10 m$^3$) with thermally insulated sides and rain-protection at the ATB (*Figure 1*).

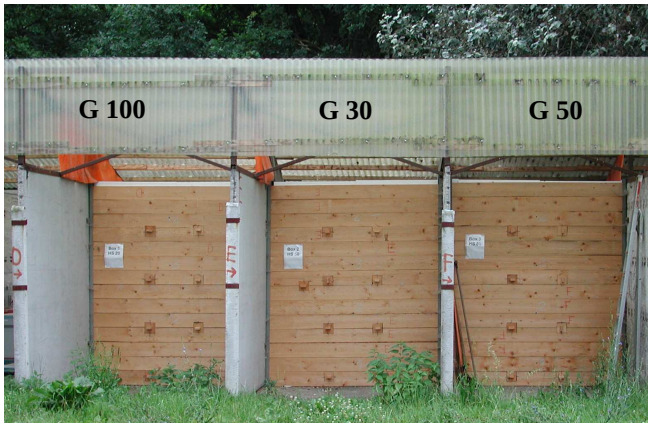



*Figure 1. Storage boxes for poplar chips and chunks of different length (G 100, G 30 and G 50)*

[1] hyphae, infested wood particles, spores

During the storage, the temperature (outside and inside of the boxes), humidity, moisture content, pH, wind speed, fungi on the wood and concentration of fungal particles in the air were measured and analysed for a storage period of 6 months (11th January 2012 – 31st July 2012).

The temperature inside of the box was measured at different heights (*Figure 2*) by an electronic pick thermometer. Ambient temperature and data outside the box were provided by the weather station of ATB.

The temperature measurements until 12th March showed a bigger fluctuation because of manual measuring. After this date an automatic measuring system was installed and air ventilation influences on the measurements have been avoided.

The moisture content, pH and the concentration of fungal particles were measured on sampling days. At the beginning of the experiments, samples were taken every two weeks after the second month of storage samples were taken only every month because many studies show that after two months the values do not change rapidly (Scholz – Idler 2001, Idler – Scholz 2001). Due to the weather conditions, measuring was not always possible. Therefore, in some cases fungi were not countable in the samples.

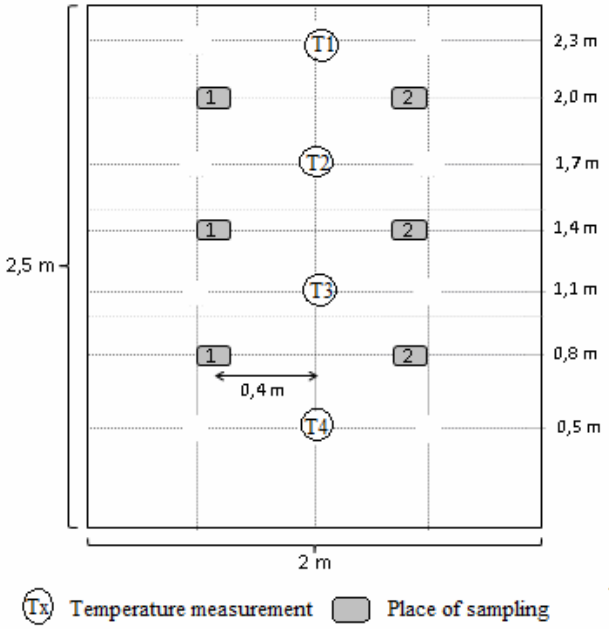



*Figure 2. Design of boxes for measuring and sampling*

Samples for the determination of moisture content and quality of the wood chips were taken from six different places per box (*Figure 2*). They were measured on a digital scale for moisture analysis immediately after sampling and were dried at 105 °C to constant weight. The moisture content was calculated from measured dry matter content. Wind speed was measured by a cup anemometer (Ahlborn, Germany).

Samples were processed to analyse pH and fungi. The pH was measured by a pH-electrode Sen Tix 41 with temperature compensation (WTW, Weilheim, Germany). The concentration of fungal particles was determined using the spread plate method. 20 g samples were shaken for 30 minutes with a rotary shaker at 180 rpm in 180 ml of Ringer solution (Merck, Darmstadt, Germany), which then was made into a decadic dilution series. An aliquot was put on special nutrient plate. Mesophilic fungi were measured by using substrate DG 18 (Oxoid, Darmstadt, Germany) and thermophilic fungi by using Malt Extract Agar (Merck, Darmstadt, Germany) with 0.01% chloramphenicol. Plates were incubated for five days at 20 °C and for two days at 37 °C. Then the number of the colonies that developed was counted and the concentration of fungal particles in a log colony forming units per g fresh mass (*lg cfu/g FM*) was

calculated. The identification was made microscopically (Pitt 1991, Samson et al. 2000, Klich 2002). The concentration of fungal particles in the air was measured directly on the top of the boxes using the Airport sampler MD8 (Sartorius, Göttingen, Germany) and the MAS 100 eco (Merck, Darmstadt, Germany). The Airport sampler MD8 (sample volume: 1000 liters, flow rate: 50 l/min), using the gelatine membrane filter method, was applied at air relative humidity under 80%. Above this value MAS 100 eco was used (sample volume: 1000 liters, flow rate: 100 l/min) which is based on the impaction principle. The control samples were taken at a height of 1.50 m. The samples of Airport MD8 were dissolved in 10 ml warm (30 °C) sterile saline solution (0.9% NaCl + 0.01% Tween 80) and were shaken for 30 minutes with a rotary shaker at 180 rpm at 30 °C. Then a decadic dilution series was made. The identification and the counting of mesophilic and thermophilic fungi were done in the same way as for the examination of fungi on the wood samples. The measurement with MAS 100 eco is a direct method. For every sampling four nutrient plates were used, two DG 18 substrates for examination of mesophilic fungi and two malt extract agar ones with 0.01% chloramphenicol for identification of thermophilic fungi. Plates were incubated for five days at 20 °C and for two days at 37 °C.

## 3 RESULTS

### 3.1 Temperature

During the examination the air temperature and the temperature in the boxes were measured at four different levels because this factor shows a close connection with development of fungi. The measurements show characteristic differences in relation to the length of the wood chips (*Figure 3*). The temperatures are given as averages of all heights from T1 to T4 (*Figure 2*).

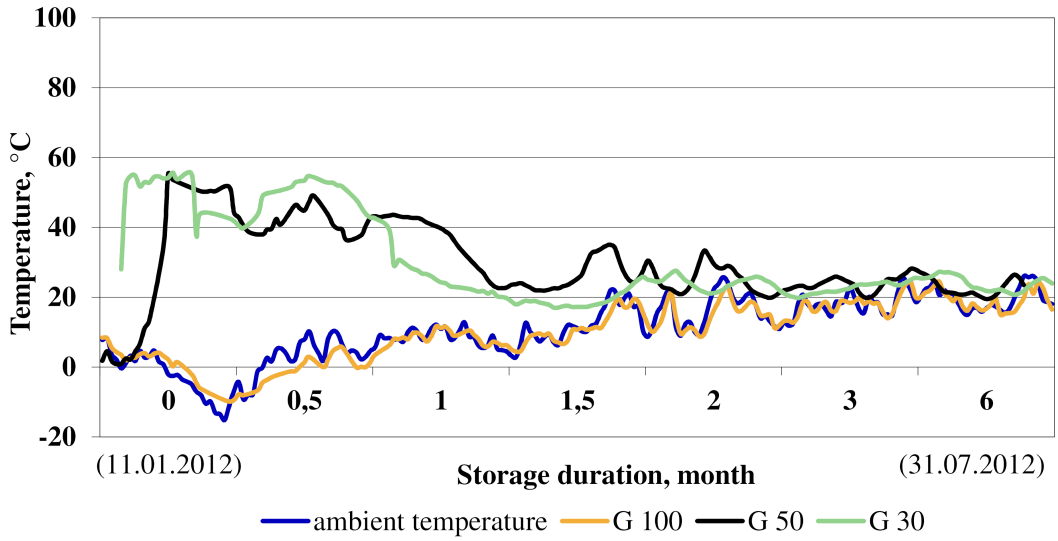



*Figure 3. Development of average temperature (°C) in the storage boxes of wood chips of different size*

Average temperatures in the box of chunks (G 100) showed a very close connection with ambient temperature. However average temperature in the box of fine (G 30) and coarse chips (G 50) rapidly increased in the first two weeks and reached their peaks at about 55 °C. Data started to drop off in the box of fine chips after storage two months and in the box of coarse chips after 10 weeks. The temperature decreased then to near to ambient temperature.

Temperatures in the box of chunks showed almost similar values at all levels. In the box of fine chips. the highest temperature was measured at the second level (T2) similar to coarse

chips *(Figure 4*). In the box of coarse chips, the temperature increased rapidly in the first of two weeks of storage and reached its temperature peaks of about 60 °C. This began to decrease after 11 weeks. Approximately 4 weeks later, the temperature started to approach the ambient temperature.

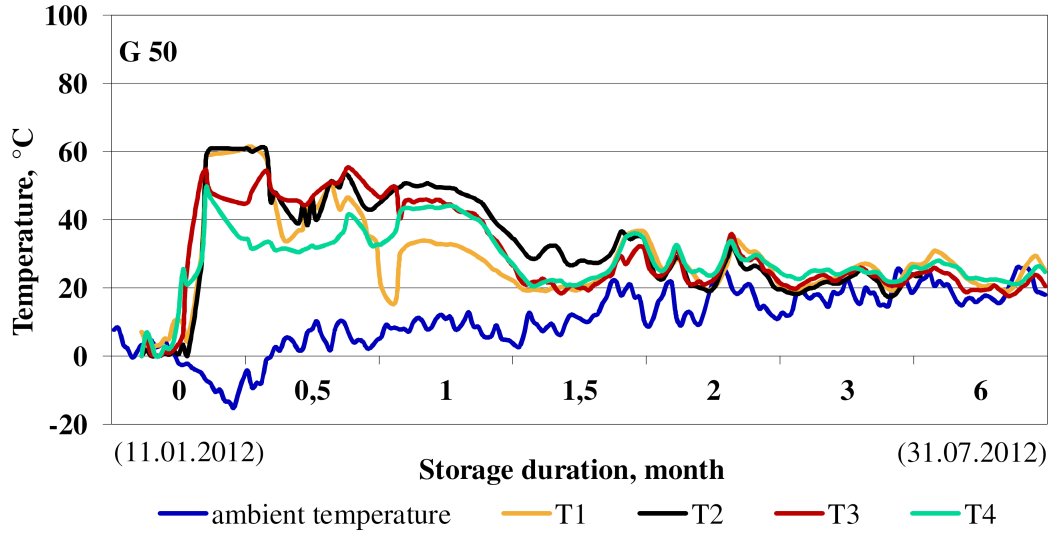



*Figure 4. Development of temperature (°C) at different heights in coarse chip piles (G 50)*

### 3.2 Moisture content, pH, fungi

In every box the same genera of fungi were found: mesophilic *Cladosporium ssp., Mucor ssp., Penicillium ssp.* and thermophilic *Aspergillus ssp.*

In the box of chunks (G 100), the number of mesophilic and thermophilic fungal particles rose after one month (*Figure 5*).

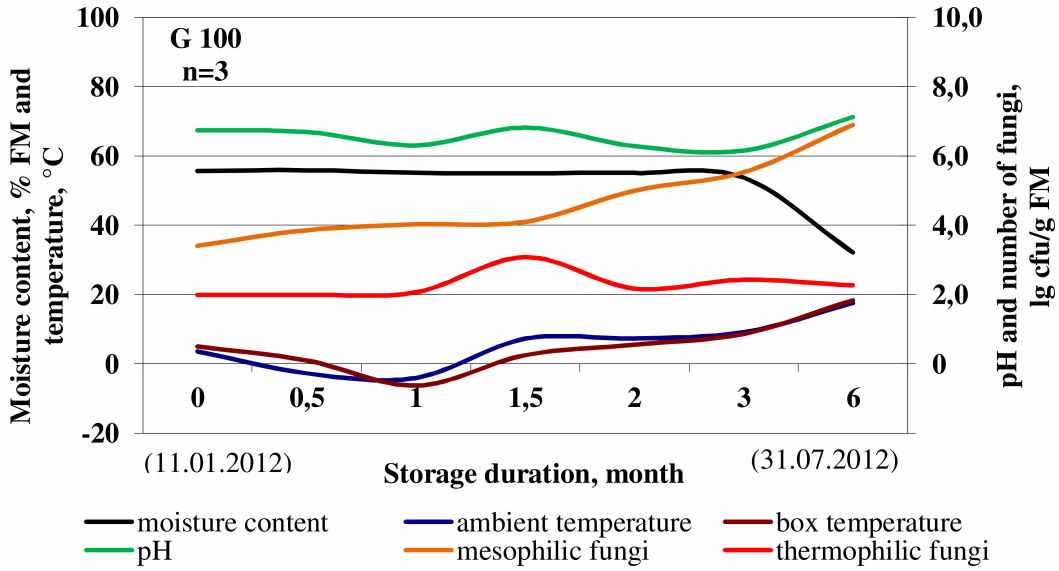



*Figure 5. Average values of moisture content (%, fresh mass), pH, temperature (°C) and development of fungi (log colony forming units per g fresh mass) during the storage of chunks (G 100)*

The quantity of mesophilic fungal particles in coarse chips (G 50) increased in one month and the order of magnitude did not change till the sixth month, while the concentration of

particles with thermophilic fungi rose rapidly (*Figure 6*). This was the tendency until the third month and then it started to sink. The moisture content decreased after the second month.

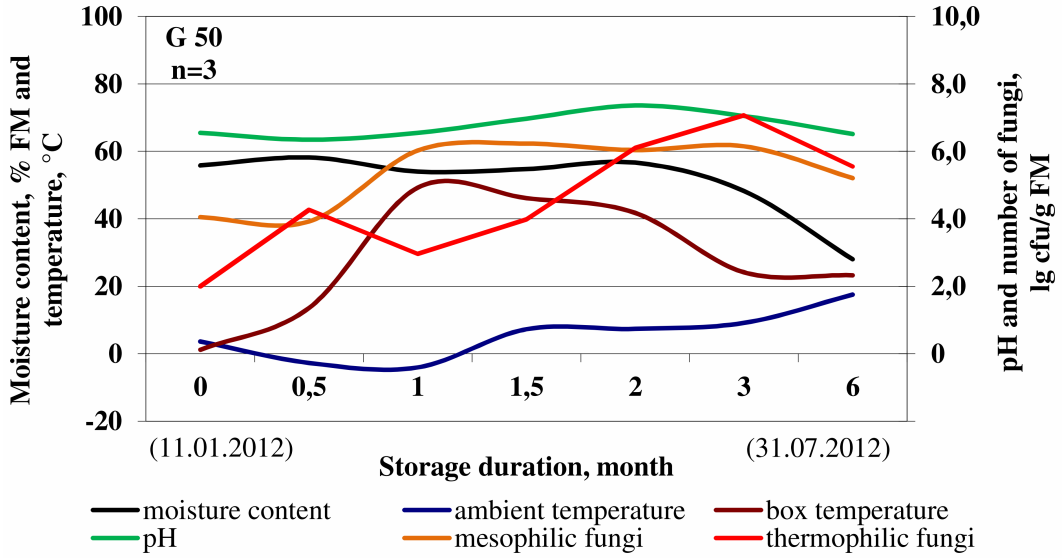



*Figure 6. Average values of moisture content (%, fresh mass), pH, temperature (°C) and development of fungi (log colony forming units per g fresh mass) during the storage of coarse chips (G 50)*

The number of mesophilic and thermophilic fungal particles in the fine chips (G 30) rose for half of a month and the order of magnitude did not change until the third month, when it started to drop (*Figure 7*). The moisture content was almost constant over the whole storage period of 6 months.

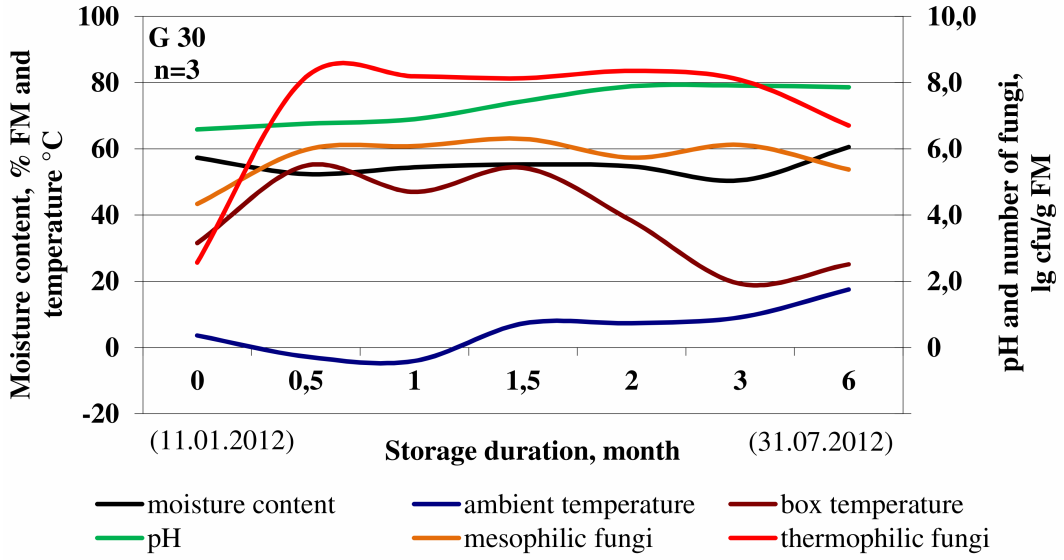



*Figure 7. Average values of moisture content (%, fresh mass), pH, temperature (°C) and development of fungi (log colony forming units per g fresh mass) during the storage of fine chips (G 30)*

In the storage box of chunks and coarse chips, the pH showed little fluctuation, but in fine chips the pH increased from 6.58 to 7.91 due to the excessive growth of fungi.

### 3.3 The concentration of fungal particles in the air

The number of mesophilic fungal particles in the air was always higher than that of thermophilic fungal particles, except in the fine chips (*Figure 8*).

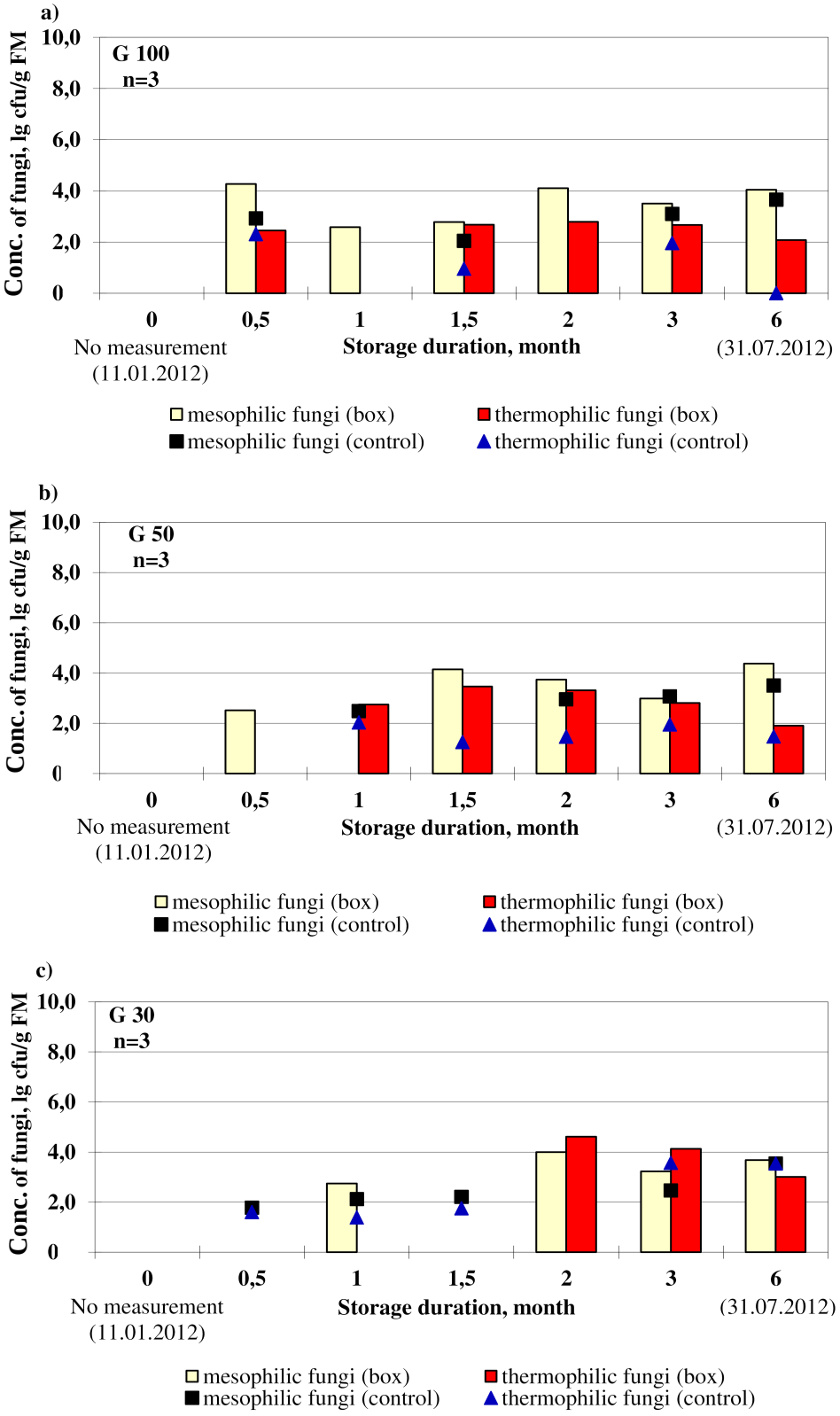



*Figure 8. Concentration of mesophilic and thermophilic fungal particles (log colony forming units per $m^3$) in the air over the storage boxes of a) poplar chunks (G 100), b) coarse (G 50) and c) fine poplar chips (G 30)*

The concentration of fungal particles in the air decreased with increasing distance. The number of mesophilic fungi was slightly higher on the top of the boxes than in the control samples and the order of magnitude was the same for all lengths of the wood chips. The number of thermophilic fungi increased until the second month of storage, and then it started to sink. Numbers grew with decreasing length of the wood chips.

The following genera of fungi were found: *Alternaria*, *Aspergillus*, *Cladosporium*, *Mucor* and *Penicillium*.

## 4 DISCUSSION AND CONCLUSIONS

Particle size is one of the most important influence factors in the storage of freshly harvested short rotation poplar in unventilated boxes. Temperatures in piles of coarse and fine chips can increase to nearly 60 °C during storage. The highest temperature was measured in fine chips. In every pile, the highest temperatures were recorded in the first period of storage; then the values start to decrease. The heating was due to the activity of microbes, residual respiration of wood cells and the restricted air flow within G 30 wood chip piles because of their lower bulk porosity. The temperatures in the box of chunks were lowest and corresponded to the ambient temperatures. This was due to the good air flow because of the coarse bulk structure.

The moisture content of chopped wood depends on the relative humidity and the temperature in the boxes. It does not fall below 45% for coarse and fine chips and reaches approximately 30% for chunks after storage for 6 months.

The pH runs from 6 to 8 and chunks have the lowest values. There is a close connection between values of pH and growth of fungi.

During storage, mesophilic and termophilic fungi develop of the genera *Alternaria, Aspergillus, Cladosporium, Mucor* and *Penicillium*.

The growing of fungi corresponds to the temperature in the box and the outside air temperature, but it does not depend on the moisture content of the wood as long as it is between 30 and 60%. The number of mesophilic fungi does not show a relationship to the length of the wood chips, but there is a close connection between the number of particles of thermophilic fungi and the length of the chips (*Figure 9*).

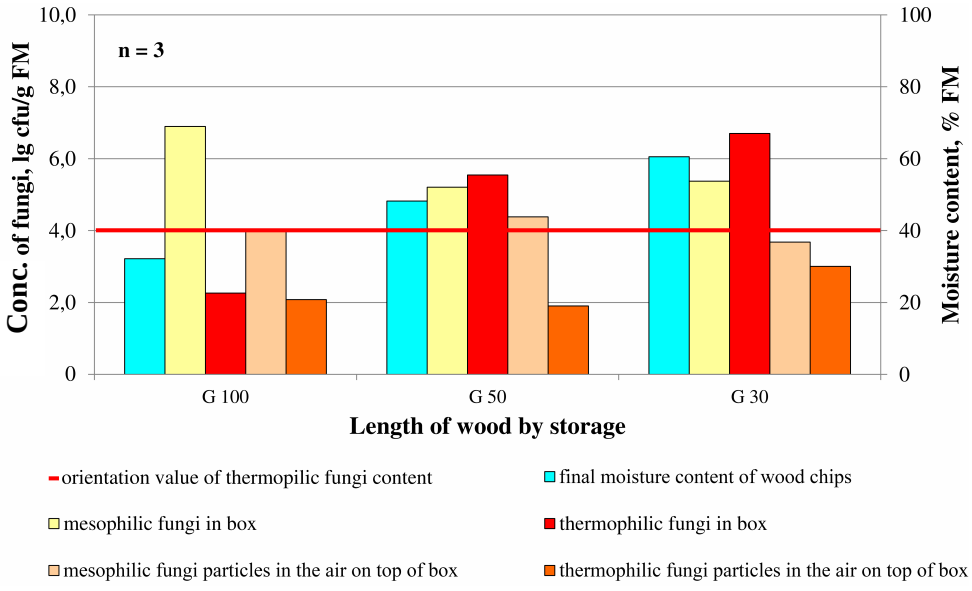



*Figure 1. Average of fungi in the box and the concentration of fungal particles in the air of chunks (G 100), coarse (G 50) and fine chips (G 30) after storage for six month*

During the examination a special focus was put on the development of thermophilic fungi, because they may be allergen, toxic and fungus pathogenic. The most important thermophilic fungus *Aspergillus fumigatus*, because it is one of the most pathogenic, can cause allergy reactions and cancer. The highest values were found in fine chips (8.4 lg cfu/g *FM*), which is one order of magnitude higher than in coarse chips, and five orders of magnitude higher than in chunks.

The development of the concentration of fungal particles in the air has a very close connection to the concentration of fungal particles in the boxes, and to the temperature in box. The number of mesophilic fungal particles in the air is related to the relative humidity of the air in the box, but this is not the case with thermophilic fungal particles in the air. The highest value of *Aspergillus ssp.* in the air was measured for fine chips (4.6 lg cfu/g *FM*), which is one order of magnitude higher than for coarse chips, and two orders of magnitude higher than for chunks. At present, there is no regulation for the limit of fungi content in the air of wood chips storages neither in Germany nor in Hungary. Therefore, the orientation value limit according to TRBA 430 (compost processing) of 4.0 lg cfu/g should be used.

Storage of coarse chips of poplar from short rotation has several advantages compared to fine chips. During the 6 months of storage the temperature in coarse chips was similar to fine chips, but the moisture content after the second month of storage is less than in fine chips, and after 6 months it reaches approximately 30%. The amount of mesophilic fungal particles in a box of coarse chips is similar to fine chips, but the number of thermophilic fungal particles is one order of magnitude less than in fine chips. This tendency was also found in fungi content in the air above the top of boxes. Storage of chunks has the most advantages, but this length of wood is not easy to handle. Compared to general form of storage fine chips, coarse chips have a better quality and are still convenient to handle. Further improvements are needed to reduce the concentration of fungal particles in the box of coarse chips as well as fungi content in the air and to speed up the drying of stored wood. Aeration with ambient or heated air could be solutions but this would increase production costs.

**Acknowledgements:** We thank Katrin Busse, Peter Kaulfuß and Werner Daries for technical assistance.

## REFERENCES

IDLER, C. – SCHOLZ, V. (2001): Storage of field wood chips-problems and solution. Renewable energy sources on the verge of the XXI century. Proceedings of the International Conference. Warsaw, 10-11th December 2001. 253–258.

IDLER, C. –SCHOLZ, V. – DARIES, W. – EGERT, J. (2004): Quality changes of chipped short rotation coppice during storage with special emphasis on human pathogen fungi. Ecological energy resources in agriculture. Proceedings of the 9th International Conference. Raudondvaris, 16–17. September 2004. 117–123.

JIRJIS, R. (1995): Storage and drying of wood fuel. Biomass and Bioenergy, 9: 181–190.

JIRJIS, R. (2005): Effects of particle size and pile height on storage and fuel quality of comminuted *Salix viminalis*. Biomass and Bioenergy, 28: 193–201.

KLICH, M.A. (2002): Identification of Common *Aspergillus* Species. Centraalbureau voor Schimmelcultures (CBS), Utrecht, The Netherlands.

ÖNORM M 7133 (1988): Energiehackgut – Anforderungen und Prüfbestimmungen. Österreichisches Normungsinstitut. Wien

PITT, J. I. (1991): A Laboratory Guide to Common *Penicillium* Species. 2nd ed. Commonwealth Scientific and Industrial Research Organization (CSIRO), Division of Food Processing. North Ryde, New South Wales, Australia

SAMSON, R. A. – HOEKSTRA, E. S. – FRISVAD, J. C. – FILTENBORG, O. (2000): Introduction to Food- and Airborne Fungi. 6th ed. Centraalbureau voor Schimmelcultures (CBS), Utrecht, The Netherlands

SCHOLZ, V. – IDLER, CH. (2001): Loss reduced and human hygienic storage of field wood chips. 1st World Conference on Biomass for Energy and Industry. Sevilla, Spain, 5-9. June 2000 : 536–239

SCHOLZ, V. – HELLEBRAND, H. J. – HÖHN, A (2004): Energetische und ökologische Aspekte der Feldholzproduktion. Bornimer Agrartechnische Berichte Heft 35. Leibniz-Institut für Agrartechnik Potsdam-Bornim e.V. : 15-32.

SCHOLZ, V. – IDLER, CH. (2005): Energieverlust und Schimmelpilzentwicklung bei der Lagerung von Feldholz-Hackgut. Bornimer Agrartechnische Berichte Heft 39. Leibniz-Institut für Agrartechnik Potsdam-Bornim e.V.

SCHOLZ, V. – IDLER, CH. – DARIES, W. – EGERT, J. (2005): Lagerung von Feldholzhackgut, Verluste und Schimmelpilzbefall. Agrartechnische Forschung 11. Heft 4. 1: 00–113.

SCHOLZ, V. – IDLER, CH. (2007): Environment-friendly storage of energy chips from short rotation coppice. Acta Technica Napocensis. Series: Applied Mathematics and Mechanics, 50, Vol (IV): 311–314.

SCHOLZ, V. – ECKEL, H. – HARTMANN, S. (2009):Verfahren und Kosten der Energieholzproduktion auf landwirtschaftlichen Flächen. Kuratorium für Technik und Bauwesen in der Landwirtschaft Schrift. : 67–80

TRBA NR. 430: Technische Regeln für Biologische Arbeitsstoffe – Verfahren zur Bestimmung der Schimmelpilzkonzentration in der Luft am Arbeitsplatz. Bundesarbeitsblatt 8/2001.

VERSITA DOI: 10.2478/v10303-012-0011-6 Acta Silv. Lign. Hung., Vol. 8 (2012) 133–144

# Investigation of the Applicability of Multi-Operational Logging Machines in Hardwood Stands

Attila László HORVÁTH[a*] – Katalin SZAKÁLOS-MÁTYÁS[a] – Béla HORVÁTH[a]

[a] Institute of Forestry and Environmental Techniques, University of West Hungary, Sopron, Hungary

**Abstract** – As a result of new developments in technology, harvesters may no longer be confined to softwood forests only. Several studies carried out in black locust, Turkey oak and beech stands have justified the use of these machines in hardwood stands. Evaluating the results of the cost and time analyses we concluded that harvesters are more efficient in several cases compared to traditional felling with chainsaws.

**harvester / logging / performance / operating cost / working day structure**

**Kivonat – Többműveletes fakitermelő gépek alkalmazhatóságának vizsgálata magyarországi lombos állományokban.** A gépfejlesztéseknek köszönhetően a harveszterek, ma már nem csak kizárólag a fenyvesekben alkalmazhatóak hatékonyan. Számos terepi méréssel sikerült Magyarországon (akác, cser, bükk állományokban) is igazolni a „gépcsodák" létjogosultságát. Az idő- és költségelemzések során kapott eredmények tudatában biztonsággal állíthatjuk, hogy a lombos állományokban is alkalmazhatóak harveszterek, sok esetben hatékonyabb munkavégzés valósítható meg, mint a hagyományos motorfűrészes fakitermelés során.

**harveszter / fakitermelés / teljesítmény / üzemóraköltség / munkaidőszerkezet**

## 1 INTRODUCTION: LOGGING WITH HARVESTERS IN HARDWOOD STANDS

Highly mechanized tree utilization, applying the harvester-forwarder combination has very rapidly become common and almost exclusive in developed West Europe (primarily in Scandinavia) and in North America where the logging of vast areas of softwood stands is common. As a result of the excellent performance and spread of these softwood harvesters, research and development began to focus on the adaptation of these machines in hardwood forests.

The operations of a harvester are identical for both hardwood and softwood stands apart from a few steps. There are no significant differences in the workflow in spite of the fact that the growth character of the trees and the structure of the forests are markedly different.

The most fundamental difference between harvesting hardwood and softwood stands is in the separation of the thick branches from the trunk which is no longer done with the pivotable gripping arms which also function as arc knives. Using the manipulator arm, the machine operator places the felling head onto the branch to be cut off. As the operator fixes the head,

* Corresponding author: ahorvath@emk.nyme.hu; H-9400 SOPRON, Bajcsy Zs. u. 4.

the limb is removed using the hydraulic chain saw equipment. After this the debranching, assorting and cross-cutting of the limbs can be carried out and the further processing of the log can be continued (Kryzanowski 2004, MacDonald – Clow 2012) (*Figure 1*).

Already in the 1970s the use of multi-purpose and multi-operational logging machines was regarded as one of the most important possibilities for increasing productivity in logging. Besides the need to increase productivity foresters recommended buying these machines in response to the increasing shortage of labour (Szepesi 1967, Csontos 1977, Horváth 2003). However, in Hungary the use of multi-purpose machines has not spead to the expected extent and rate, for the following reasons:

- As the machines were obtained from various vendors, it turned out to be extremely difficult to achieve optimal coordination of this equipment when they were assembled into machine lines
- In the earlier economic and organisational environment a 4-5-fold cost increase appeared compared to traditional logging with chain saws.

## 2 ANALYSIS AND EVALUATION OF THE WORK WITH HARVESTERS

Nowadays multi-operational logging machines work in both hardwood and softwood stands (*Figues 2-5*). Depending on conditions of terrain, some of this equipment is primarily used in hardwood stands, while other types are almost exclusively used in softwood forests. Based on the foreign results in efficiency and productivity of harvesters, there is no doubt that these machines should be the first choice for domestic softwood forests too. However, their use in domestic hardwood stands raises many questions. In the past few years multi-operational logging machines have already been used for logging domestic forests (in black locust, alder, hybrid poplar, Turkey oak, hornbeam-oak, beech, hornbeam-Scots pine, spruce, Scots pine and black pine stands). Clearcutting, thinning, preparatory cutting and sanitary cutting were all done with these machines.


*Figure 1. Harvester logging of a Turkey oak stand*

In order to analyze and evaluate the structure of the work day and performance of the harvesters, field surveys were done with continuous time measurement. Besides recording the duration of individual actions, the total volume of the timber processed in each cycle and the distances of changeovers were also recorded. During the survey, the following types of actions were distinguished:

- Grabbing the tree (*GT*): equals the time required for the machine operator to place the harvester head onto the base of the tree, using the manipulator arm.
- Felling and processing the log (*F*): involves the time for felling, preassembly, debranching of the log as well as the conversion into assortments and piling by assortments.
- Changeover (*C*): is the duration of machine displacement.
- Felling only (*FO*): time spent on logging very thin or poor quality (e.g. completely rotten) logs, which do not yield valuable assortments.
- Arranging branch material (*B*): transfer and rearrangement of branches obstructing the path of logging.
- Arranging timber (*T*): transfer and arrangement of timber stacks obstructing the path of logging.
- Rest period (*R*): time for meeting personal needs.
- Troubleshooting (*TH*): time for fixing technical defects in the machinery.
- Waiting (*W*): other time losses (e.g. phone calls)

*Figure 2. Valmet 911.3 harvester, Szentgál*

*Figure 3. Ponsse HS16 Ergo harvester, Balinka*

*Figure 4. Timberjack 1270B harvester, Horvátzsidány*

*Figure 5. Ponsse Buffalo Dual harwarder, Kecskemét*

## 2.1 Clearcutting in a black locust stand using Valmet 911.3 and Silvatec 896 TH-H harvesters

A field survey was made in a black locust stand, group-mixed with 25% black pine, with a total area of 23.6 hectares. 4.3 hectares of the stand with pure black locust forest were clearcut. The average age was 42 years, average stem height was 17 m, average diameter at breast height measured 20 cm. With a canopy closure of 96%, there was an average of 640 stems/hectare, with a standing volume of 148 $m^3$/hectare. This row-planted seedling stand was classified into yield class IV. The shrub layer was moderate (30–70%) but even. The logging and processing of the trees was done with the Valmet 911.3 and Silvatec 896 TH-H harvesters. During clearcutting, the machines all worked in 15 m wide tracts, moving periodically (logging changeover). The distance of changeovers ranged between 2-10 m, depending on the location of the trees within the tracts. The branches of the shrub layer, (which had been previously cut with chain saw), and the tree branches were continuously moved to the right side of the tract by the machine operator. The branchwood was collected at the right side of the skidding trail (located in the middle of the tract) while different groups of roundwood assortments were collected at the left side. Two types of assortments were produced: firewood and pole wood. Pole wood was converted from the logs with a length of 3 m and a top diameter of 15–25 cm. The rest of the roundwood was sorted for fire wood, also with a length of 3 m. The forwarding of the timber was done with a Valmet 860.3 machine.

The field survey and data recording of the Valmet harvester lasted 698.47 minutes (2 days). During the period of the measurement (*Figure 6*) 41.3% of the operating time was spent on felling trees, 3.9% was for changeovers, and around 22.9% of the time was spent arranging the branch material. Altogether 37 $m^3$ timber was harvested. The shift performance and performance per hour was determined on the basis of the work day structure and the volume of logged timber. Using the country specific machine utilization factor ($P$ = 60% for Hungary) expected performances were calculated (*Table 1*). The productive time ($P$) of the harvester during the investigation was 80.1%.

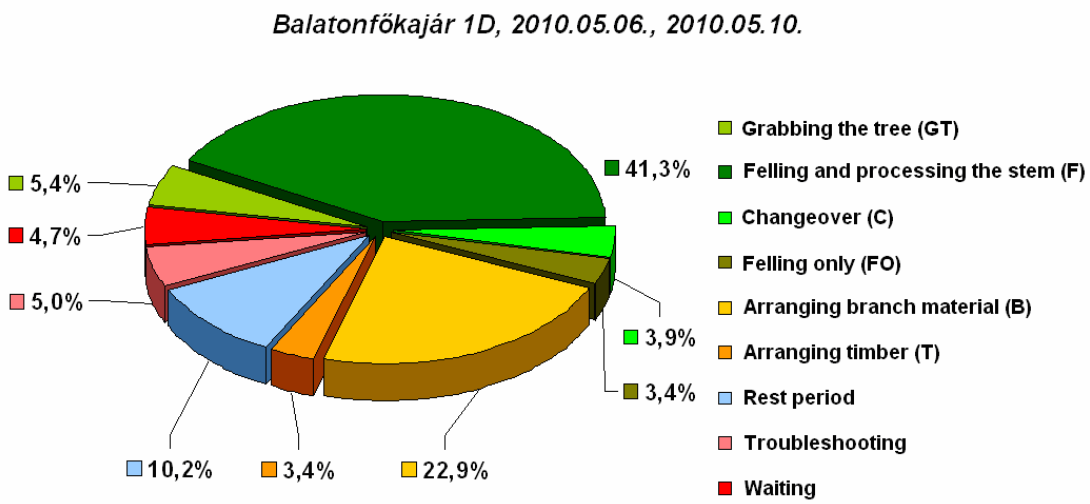



*Figure 6. Operating time structure of the Valmet 911.3 harvester*

*Table 1. Performance of the Valmet 911.3 harvester*

| **Performance** | | $m^3$/h | $m^3$/shift |
|---|---|---|---|
| Logging | (*GT+F+C*) | 6.3 | 50.3 |
| Logging + site clearing | (*GT+F+FO+C+B+T*) | 4.0 | 31.7 |
| During the whole measurement time | (*S*) | **3.2** | **25.4** |
| **Expected performance** (P=60%) | | | |
| Logging | (*GT+F+C*) | 3.8 | 30.2 |
| Logging + site clearing | (*GT+F+FO+C+B+T*) | **2.4** | **19.0** |

The field survey and data recording of the Silvatec harvester took 263.87 minutes, and 12.4 $m^3$ of timber was harvested. During the period of the measurement 43.0% of the operating time was spent on tree felling and processing, and 29.5% was used for arranging the branch material (*Figure 7*). The achieved and expected performance data are summarized in *Table 2.*

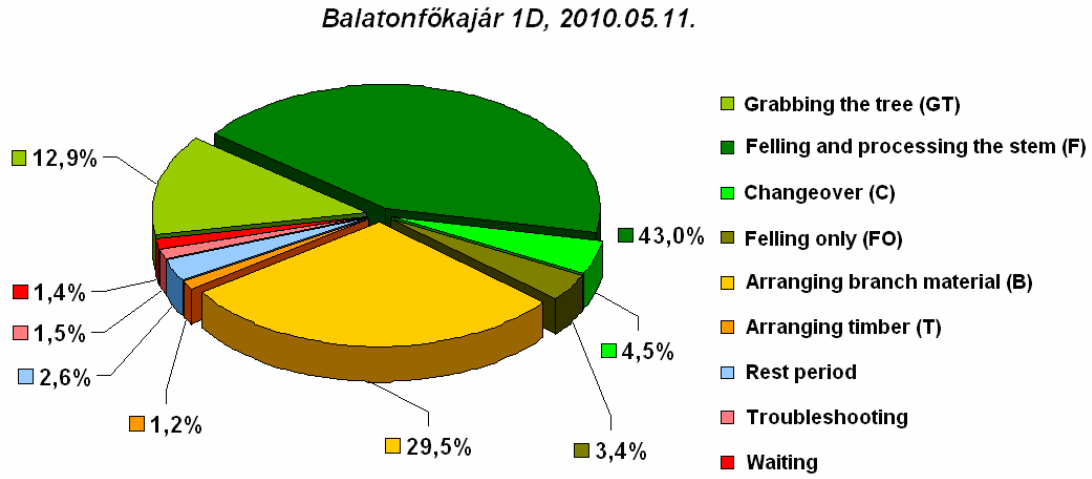



*Figure 7. Working time structure of the Silvatec 896 TH-H harvester*

The low performance of the machines (3.2 $m^3$/h and 2.8 $m^3$/h) can be primarily attributed to the weak yield class of the stand and to the fact that the heads of the harvesters had sizes at the lower application limit of the economical range. Moreover, efficient work was also hindered by the fact that machine operators were not sufficiently experienced.

*Table 2. Performance of the Silvatec 896 TH-H harvester*

| **Performance** | | $m^3$/h | $m^3$/shift |
|---|---|---|---|
| Logging | (*GT+F+C*) | 4.7 | 37.3 |
| Logging + site clearing | (*GT+F+FO+C+B+T*) | 3.0 | 23.9 |
| During the whole measurement time | (*S*) | **2.8** | **22.6** |
| **Expected performance** (P=60%) | | | |
| Logging | (*GT+F+C*) | 2.8 | 22.4 |
| Logging + site clearing | (*GT+F+FO+C+B+T*) | **1.8** | **14.3** |

## 2.2 Thinning in a mixed hornbeam – Turkey oak – beech stand using the Ponsse HS16 Ergo harvester

The increment thinning in a 72-year-old, 14.7 hectare total area of hornbeam – Turkey oak – beech mixed stand was done using a Ponsse HS16 Ergo multi-operational logging machine. According to the data of the management plan, the average diameter of the stand was 21 m, and average diameter at breast height was 26 cm. Before the thinning, workers of the forest management unit had marked the trees intended for cutting with paint. The machine operator converted the logs into assortments as follows: from a top diameter of 25 cm, 3 m logs, from a top diameter of 15 cm, 2.5 m industrial firewood, and 4 m household firewood. The timber was forwarded using a Timberjack 1110 forwarder. The cross-cutting of the 4 m household firewood assortment into 1 m chunks was done at the landing site using chain saws.

During the study (nearly 4 hours) 48.4 $m^3$ timber were produced. About 60.0% of the working time was spent on grabbing, felling and debranching, conversion into assortments, cross-cutting, and stacking (*Figure 8*). This type of harvesting involves a relatively high frequency of changeovers. During the experiments, 91 changeovers were made, with an average distance of 13 m. The time loss of over 20% (46.3 minutes) could be attributed to the malfunctioning of a sensor in the cutting unit of the harvester head.

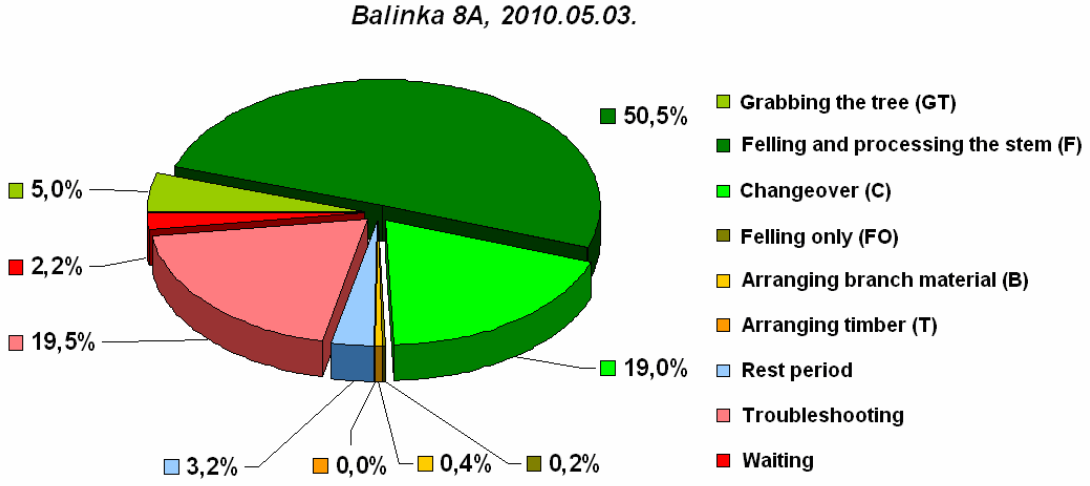



*Figure 8. Working time structure of thinning operation*

Despite the relatively high proportion of time losses, 75.2% of the total working period was productive. The performance per hour calculated for the whole time was 12.3 $m^3$/h (*Table 3*). The performance of the shift during the productive periods exceeded 130 $m^3$/shift. Under similar conditions, the expected performance of the machines was calculated at 9.8 $m^3$/h.

*Table 3. Measured and expected performance values*

| **Performance** | | $m^3$/h | $m^3$/shift |
|---|---|---|---|
| Logging | (*GT+F+C*) | 16.5 | 132.0 |
| Logging + site clearing | (*GT+F+FO+C+B+T*) | 16.4 | 130.9 |
| During the whole measurement time | (*S*) | **12.3** | **98.4** |
| **Expected performance** (P=60%) | | | |
| Logging | (*GT+F+C*) | 9.9 | 79.2 |
| Logging + site clearing | (*GT+F+FO+C+B+T*) | **9.8** | **78.5** |

### 2.3 Selective regeneration cutting in a Turkey oak and beech stand using the Valmet 911.3 harvester

The whole of the 8.5 hectare Turkey oak – beech stand was subject to the selective regeneration cutting. The average age of the stems was 77 years, average tree height was 19 m and the average diameter at breast height was 27 cm. With a canopy closure of 70%, there was an average of 410 stems/hectare, with a standing volume of 231 $m^3$/hectare. During the logging the rubber-tyred Valmet 911.3 harvester had to cope with a shrub layer of over 70% and with the large amount of Turkey oak regrowth. The machine discontinuously moved forward in a 15-20 m wide tract, while forming a winding skidding trail in front of itself driving around remaining trees. The stacking of the 3-meter-long assortments was done beside the machine, the branch material, thin twigs and the limbs of the crown (diameter under 5 cm) were collected at the right and left sides of the skidding trail. The shrubs, which were a hindrance to the logging process and had been previously cut by chain saws, were collected by the machine operator to the near side of the skidding trail using the manipulator arm of the harvester. The roundwood assortment was forwarded with a Valmet 860.3 machine. In order to protect the tender tree regrowth and to ensure undisturbed development, both harvester and forwarder machines moved along the skidding trails and the branch material as well as the previously chain-sawed shrubs were both collected next to the these pathways.

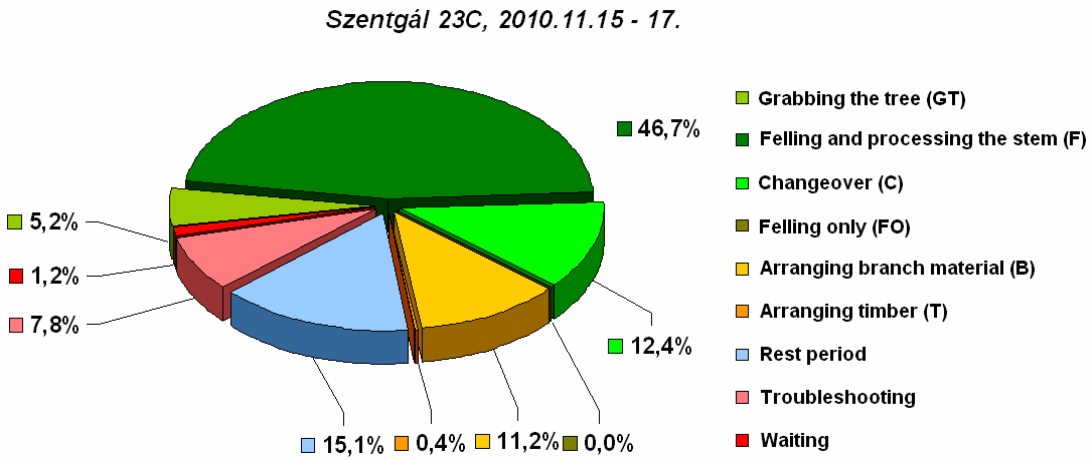



*Figure 9. Work time structure of regeneration cutting*

The field survey and data collection were done in 3 days (1130 minutes). The percentage of the actions during the working time is depicted in *Figure 9*. 75.9% of the overall working hours were productive. The proportion of troubleshooting, waiting and rest periods is quite low.

The volumes of harvested and processed timber (196.1 $m^3$) and the duration of the single actions were considered to estimate performance values. The hourly performance of the Valmet 911.3 multi-operational logging machine (related on the whole measurement time) was 10.4 $m^3$/h. The expected shift performance – under similar circumstances and stand conditions – was calculated as 65.6 $m^3$/shift (*Table 4*). The average changeover distance was 10 m, and the average time needed for changeovers 0.35 min.

*Table 4. Measured and expected performance values in the preparatory cutting*

| **Performance** | | $m^3$/h | $m^3$/shift |
|---|---|---|---|
| Logging | (*GT+F+C*) | 16.2 | 129.2 |
| Logging + site clearing | (*GT+F+FO+C+B+T*) | 13.7 | 109.4 |
| During the whole measurement time | (*S*) | **10.4** | **83.0** |
| **Expected performance** (P=60%) | | | |
| Logging | (*GT+F+C*) | 9.7 | 77.5 |
| Logging + site clearing | (*GT+F+FO+C+B+T*) | **8.2** | **65.6** |

The performance data indicate that harvesters can be used efficiently for logging Turkey oak stands, despite their strong branch system and the great number of warped and twisted logs. Optimal workflow and proper machine design led to minimum site disturbance and damage to the remaining stand and regrowth caused by the machine-assisted logging.

### 3.4 Sanitary cutting in a beech stand struck by windthrow using a Valmet 911.3 harvester

Investigations were carried out in a 60-year-old storm damaged stand. The tree species were beech, common ash, hornbeam and Turkey oak with an average height of 12–14 m, and with an average diameter at breast height of 13–17 cm. Storm damage affected primarily trees with large crowns and large diameters ($d_{1,3}$: 25–40 cm). Windthrow hit the trees in groups, uprooting them as a result of heavy and intense spring rain. In most cases, trees fell onto each other or got stuck on remaining standing trees. Following the rules applied to the exploitation of storm-damaged stands, the logging of the thrown trees was carried out moving from the outside inwards and from the top downwards. In some cases the special position of the trees required an alternate processing to commence at the crown and moving towards the stump of the tree. The 3 m long assortments were forwarded with the Valmet 860.3 machine.

Data recording of the field survey took place in Szentgál 63B forest subcompartment for 312 minutes. Most of the working time was used for felling and processing, changeovers and arranging the branch material (*Figure 10*).

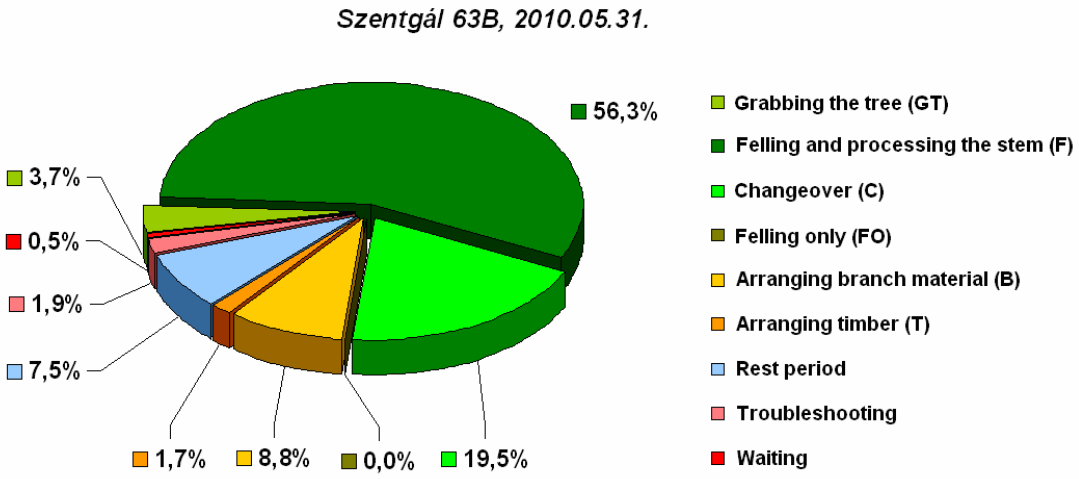



*Figure 10. Working time structure of sanitary cutting in a storm-damaged stand*

Changeovers were done in two different ways. One type of changeover was done by moving between the storm-damaged spots (long distance changeovers); the other type of movement (short distance changeovers) was done when timber processing was finished (*Figure 11*). The relatively high scatter of data was due to field conditions, the start time acceleration and the stop time deceleration.

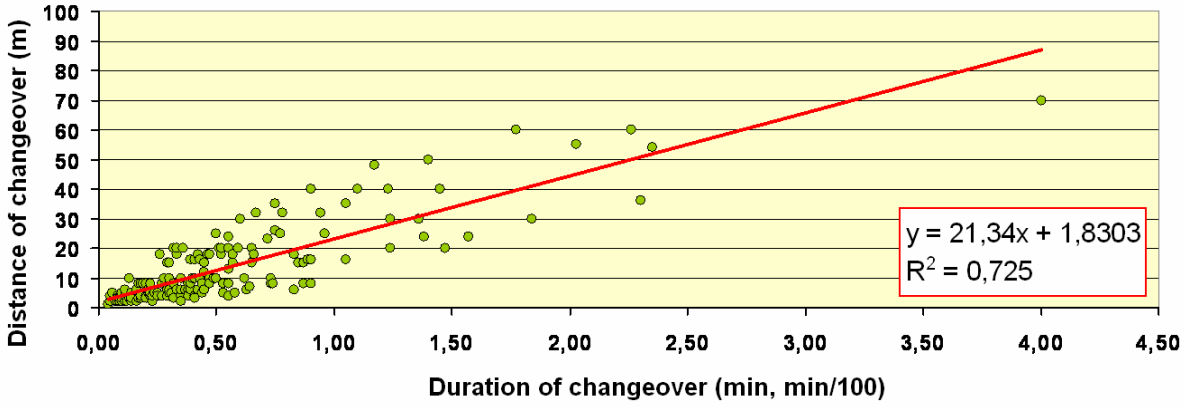



*Figure 11. Relation between the distance and the duration of changeovers*

Logging storm-damaged stands is always about saving valuable timber. This also involves an above average number of dangerous and risky situations. Following the rules of dangerous logging is recommended for both motor-manual, and mechanized harvesting. Machine operators are safer compared to their colleagues doing the same jobs with a chain saw. Nevertheless careless work can cause dangerous accidents as well as severe and costly technical breakdowns. The effect of additional care required by the special circumstances on performance is negligible (*Table 5*), because the machine operator can manage the processing, moving and cross-cutting of the logs from a safe distance using a crane. The minor decrease in performance can be attributed to the frequent, long distance changeovers.

*Table 5. Measured and expected performances*

| **Performance** | | $m^3$/h | $m^3$/shift |
|---|---|---|---|
| Logging | (*GT+F+C*) | 10.0 | 79.8 |
| Logging + site clearing | (*GT+F+FO+C+B+T*) | 8.8 | 70.4 |
| During the whole measurement time | (*S*) | **7.9** | **63.4** |
| **Expected performance** (P=60%) | | | |
| Logging | (*GT+F+C*) | 6.0 | 42.3 |
| Logging + site clearing | (*GT+F+FO+C+B+T*) | **5.3** | **63.4** |

## 3 ECONOMIC ASPECTS OF THE USE OF HARVESTERS

When considering the utilization of machines with high investment costs, the operating costs and the specific costs of logging are always key factors. The operating costs were estimated using the following formula (the meaning of the characters is shown in *Table 6*):

$$k_B = \frac{A * a}{J * 100} * (1 + r) + \frac{A * p}{2 * J * 100} + B_f * b_j + K_E + \frac{u * A_u * (1 + \frac{o}{100}) * P}{100} \quad (1)$$

Based on this formula the operating costs of a Valmet 911.3 harvester, which logged a black locust stand, have been determined (*Table 6*).

*Table 6. Operating costs of a Valmet 911.3 harvester (2010)*

| **Tag** | | | **Valmet 911.3** | **unit** |
|---|---|---|---|---|
| *A* | Acquisition cost (w/o VAT) | | 60 | M Ft |
| *n* | Machine life (amortization time) | | 7 | yr |
| *J* | Operating hours per annum | | 2 000 | h/yr |
| *a* | Amortization factor *(100/n)* | | 14.29 | % |
| *r* | Reparation proportion | | 1.20 | |
| *p* | Rate of interest | | 5.25 | % |
| *u* | Fuel consumption | | 10.00 | l/h |
| $A_u$ | Fuel cost | | 320 | Ft/l |
| *o* | Lubricant cost proportion | | 45 | % |
| *P* | Utilization (Productive hours / Operating hours) | | 60 | % |
| *e* | Other costs / operating costs with wage ratio | | 0.40 | % |
| $B_f$ | Paid wages | | 800 | Ft/work. h |
| $b_j$ | Multiplier for wage expenses | | 1.29 | |
| $K_a$ | Amortization costs | $A/(n*J)=A*a/(J*100)$ | 4 286 | Ft/work. h |
| $K_r$ | Maintenance and repair costs | $r*K_a$ | 5 143 | Ft/work. h |
| $K_p$ | Cost of interest | $(A*p)/(2*J*100)$ | 788 | Ft/work. h |
| *F* | Operational costs per productive hour | $u*A_u*(1+o/100)$ | 4 640 | Ft/work. h |
| $K_F$ | Operational costs | $F*P/100$ | 2 784 | Ft/work. h |
| $K_B$ | Wage cost | $B_f*b_j$ | 1 032 | Ft/work. h |
| $O_1$ | Altogether | $K_a+K_r+K_p+K_F+K_B$ | 14 033 | Ft/work. h |
| $K_E$ | Other costs (tax, storage, insurance) | $O_1*e/100$ | 56 | Ft/work. h |
| ***k*$_B$** | **Operating costs with wage** | $\mathbf{K_a+K_r+K_p+K_F+K_B+K_E}$ | **14 089** | **Ft/work. h** |

The specific costs of the investigated loggings are summarized in *Table 7.* According to the data obtained from the surveyed forest management units, the costs of traditional logging (felling with chain saws and using cable skidders) were also evaluated (*Table 8*) for the same forest stands. The calculations demonstrate that the cost of logging with harvesters is between 2,000–3,000 Ft/m$^3$, forwarding costs 2,500–3,500 Ft/m$^3$. Thus the total costs of logging using whole process mechanization in the investigated hardwood stands are between 4,500–6,500 Ft/m$^3$ Although traditional logging is more cost-effective than mechanized logging, work with harvesters is much more productive, and includes site clearing. Felling site harvesting losses can easily be utilized (e.g. with chipping), therefore the cost difference might be even smaller. Another advantage of using the combination of a harvester and a forwarder is less site disturbance which means also less impact on the environment and the ecosystem.

*Table 7. Logging with harvesters*

| **Harvester** | **Performance** $m^3/h$ | **Operating cost** Ft/h | **Specific cost** $Ft/m^3$ |
|---|---|---|---|
| Clearcutting | 3.2 | 14089 | 4403 |
| Increment thinning | 12.3 | 13031 | 1059 |
| Regeneration cutting | 10.4 | 14395 | 1384 |
| Windthrow | 7.9 | 14351 | 1817 |
| Average of logging with a harvester | | | 2166 |
| Forwarding | | | 2500 |
| **Average** | | | **4,666** |

*Table 8. Traditional logging*

| **Chain saw, cable skidder** | **Performance** $m^3/h$ | **Specific cost** $Ft/m^3$ | |
|---|---|---|---|
| Clearcutting | 2.1 | 4000 | 4500 |
| Increment thin. | 3.1 | 3200 | 3700 |
| Regeneration c. | 3.3 | 2800 | 3200 |
| Windthrow | 2.9 | 3500 | 3700 |
| **Average** | | | **3,575** |

## 4 SUMMARY

Regarding optimal technological developments, logging done with harvesters can also be introduced to hardwood stands. This however requires higher level professional attention and control.

The 3-year ”forstINNO” project starting in 2006 – in which the Institute of Forest and Environmental Techniques has also participated – primarily targeted the ecologic, economic and technical evaluation of the workflow of a harvester, specially designed for hardwood stands in 9 European countries (ranging from England to Lithuania). Based on the data measured in five countries and according to their own research, it was concluded that with trees averaging 0.07-0.41 $m^3$, the productivity related to productive time ranges between 4.9–16.4 $m^3/h$, also taking into account the different levels of difficulty during working operations and the number of pieces in an assortment (Erler 2007). Based on the operating costs and performance data of a harvester, costs could be calculated, which did not differ significantly from the charges of a small logging business operating at the motor-manual level.

The results of the international project are in accordance with the conclusions of the present study. These findings are also confirmed by the performance data of the harvesters operating in Hungarian hardwood stands, even if some of professionals still look with mistrust at the utilization of multi-operational logging machines in hardwood stands.

**Acknowledgements:** The European Union and the European Social Fund have provided financial support for the project under the grant agreement no. TÁMOP-4.2.2.B-10/1-2010-0018.

## REFERENCES

CSONTOS, GY. (1977): A műszaki fejlesztés és a kutatás kapcsolata. [Relationship between technical development and research] Az Erdő, OEE, Budapest, XXIV (4): 67–170. (in Hungarian)

ERLER, J. (2007): Development of Ecologically Compatible, Highly Productive Methods of Timber Harvesting of Central European Forestry. Scientific Part forstINNO, Dresden. 23. p.

HORVÁTH, B. (2003): Erdészeti gépek. [Forest machinery] Szaktudás Kiadó Ház, Budapest. 296. p. (in Hungarian)

SZEPESI, L. (1976): Az ipari erdőkben alkalmazható géprendszerek kialakítása. [The performance of employable machine systems in the industrial forests] Az Erdő, OEE, Budapest. XXV. (8): 352–354 (in Hungarian)

KRYZANOWSKI, T. (2004): Quebec equipment innovators. The Logging and Sawmilling Journal, September 2004. http://www.forestnet.com/archives/Sept_04/in_the_woods_innovations.htm

MACDONALD, P. – CLOW, M. (2006): The industrialization of tree harvesting systems in the Eastern Canadian forest, 1955–1995.(Publishing company?) Canadian Committee on Labour History, Labour / Le Travail 58:145–167

http://www.historycooperative.org/journals/llt/58/macdonald.html

VERSITA DOI: 10.2478/v10303-012-0012-5 Acta Silv. Lign. Hung., Vol. 8 (2012) 145–155

# Photodegradation of Timber of Three Hardwood Species Caused by Different Light Sources

László TOLVAJ* – Dénes VARGA

Institute of Physics and Electrotechnics, University of West Hungary, Sopron, Hungary

**Abstract** – In this study, resistance of black locust, beech and poplar wood to photodegradation was tested, applying sunlight, a xenon lamp and a mercury vapour lamp. The irradiation time was 200 hours for sunlight and the xenon light and 20 hours for the mercury light. The changes were monitored by colour measurements and infrared spectroscopy. The colour change of black locust was more intensive at the beginning of the irradiation than that of the beech and poplar. The degradation of aromatic structure of lignin (absorbing at 1510 and 1596 $cm^{-1}$) in black locust was minor compared to the same changes of beech and poplar during the first 10 hours. The mercury lamp induced more intensive changes both in colour and in infrared spectrum than the other two light sources. The results show that the high extractive content of black locust absorbs a considerable amount of light radiation protecting the main chemical components of wood.

**colour change / infrared spectrum / mercury lamp / photodegradation / xenon lamp / wood**

**Kivonat – Három lombos fafaj faanyaga fotodegradációs tulajdonságainak összehasonlítása különböző fényforrások alkalmazása esetén.** Akác, bükk és nyár faanyagok fotodegradációval szembeni ellenálló képességét vizsgáltuk napsugárzás, xenon lámpás és higanygőz lámpás besugárzás esetén. A kezelési idő 200 óra volt a napsugárzásos és a xenon lámpás besugárzásnál, és 20 óra a higanygőz lámpás besugárzásnál. A változásokat színméréssel és az infravörös spektrum felvételével követtük nyomon. A kezelés kezdetén az akác színváltozása sokkal intenzívebb volt, mint a bükké és a nyáré. Az első 10 órában a lignin aromás gyűrűjének degradációja (abszorpciós helyei: 1510 és 1596 $cm^{-1}$) akác esetében sokkal kisebb volt, mint bükk és nyár esetében. A higanygőz lámpás kezelés sokkal intenzívebb változást produkált (a színváltozásban és az infravörös spektrumban is), mint a másik két fényforrás. Az eredmények azt mutatják, hogy az akác magas extraktanyag tartalma elnyeli a fénysugarak jelentős részét, ezzel megvédve a faanyag fő kémiai összetevőit.

**színváltozás / infravörös színkép / higanygőz lámpa /fotodegradáció /xenon lámpa / faanyag**

## 1 INTRODUCTION

The colour inhomogeneity of wood is one of the most beautiful creations of nature. The colour hue of wood varies between red and yellow ensuring a warm and pleasant effect. This colour harmony of wood is however sensitive to light and heat. The combination of light and

* Corresponding author: tolla@fmk.nyme.hu; H-9400 SOPRON, Bajcsy Zs. u. 4.

heat treatment results in an even more intensive discoloration than separately (Mitsui et al. 2001, Mitsui 2004, Mitsui et al. 2004, Mitsui – Tsutchikawa 2005, Mitsui et al. 2005). The colour change of wood is the most sensitive indicator for the degradation caused by sunlight. The colour alteration of black locust wood can be seen after 2–3 hours of irradiation by the naked eye. The objective colour monitoring of photodegradation has been applied in wood science only for the last twenty years (Tolvaj – Faix 1995, Kawamura et al. 1996, Pastore et al. 2004, Pandey 2005, Oltean et al. 2009). For this purpose, the CIE $L^*$ $a^*$ $b^*$ colour co-ordinate system has been used in most cases.

Sunlight is the main factor that causes the greatest changes in the surface properties of wood during outdoor exposure. Careful investigation of this type of degradation of wood is difficult with outdoor exposure because weather conditions are neither controllable nor repeatable. Therefore the light-induced degradation of wood is usually investigated under artificial conditions. The most frequently used artificial light source is a xenon lamp. As the ozone layer in the Earth's stratosphere is constantly dwindling, nowadays more ultraviolet (UV) radiation reaches the Earth's surface than before. Therefore the UV B wavelength region (280-315 nm) has to be taken into consideration. Xenon lamps have no emission in the UV B region while the emission spectrum of a mercury lamp has wider range of UV light (Tolvaj – Mitsui 2005).

Comparative studies on the effect of sunlight and artificial light sources in the weathering of wood can hardly be found in the literature. Ota et al. (1997) tested the colour stability of acetylated veneers of kiri (*Paulownia tomentosa* Steud.) irradiated by sunlight and light generated by mercury lamp. Podgorski et al. (1996) measured the effect of outdoor and artificial weathering of coated wood by glass transition temperature. The maximum value of glass transition temperature was found to be about 24°C. The cycles of treatment consisted of plunging samples into distilled water, drying and light irradiation with UV lamps. Pandey and Vuorinen (2008) carried out a detailed study of photodegradation of wood surfaces using a xenon lamp and a UV laser emitting at 244 nm. The UV resonance Raman spectra of laser irradiated wood showed similar behaviour as the wood irradiated by xenon light, i.e. overall broadening and a rapid reduction in the intensity of the lignin aromatic structure. The degradation rate caused by laser was very high. However, the extent of band broadening was higher in wood irradiated by xenon light indicating the generation of several different types of structures as compared to laser irradiation which produced only one type of structure.

Black locust wood has a high extractive content. This feature determines its high resistance to fungal attacks (Molnár – Bariska 2002), Pandey (2005) compared to the photo-discoloration of natural and extractive free wood samples. The extractive free specimens exhibited a monotonous increase in colour change with increasing irradiation time. Unextracted wood surfaces showed a rapid colour change during the initial period of exposure which decreased after a prolonged exposure time. Analysis of colour changes and FTIR spectra measured on irradiated wood surfaces indicate that the presence of extractives increases the rate of photo-discoloration and results in an apparent increase of the delignification rate of wood surfaces in the initial period of exposure. The apparently increased rate of delignification in unextracted wood has been explained on the basis of photo-degradation of polyphenolic extractives.

In our experiments, the samples were exposed to sunlight only on sunny days to determine the effect of sunlight alone. The objective of this work was to compare the photodegradation effects of black locust, beech and poplar wood caused by natural and two different artificial light sources. Another aim of this study was to find a test method which can simulate the changes in wood surface discolouration caused by sunlight.

## 2 MATERIALS AND METHODS

The investigated hardwood species were: black locust (*Robinia pseudoacacia*), beech (*Fagus crenata*) and poplar (*Populus canescens*). The species were chosen because of the differences in extractive content. Black locust has a high extractive content (4–9%) while poplar has hardly any extractives. Black locust is often used in outdoor applications. Poplar species are sometimes used as structural wood. Beech has medium extractive content and it often shows unwanted discolouration in furniture due to sunlight, even in indoor use. Planed surfaces with a tangential orientation were prepared. The tangential surface was chosen to reduce the colour inhomogeneity of the surface. The sample size was 50x10x2 $mm^3$. The samples of different series were prepared from the same board. All species were represented by a series of 2 samples, and 5 points of fixed location were measured. The data presented in this work are the average of 10 measurements. The absolute value of the data is not important for these investigations. Since only the tendency of change is important, no statistical analysis was made. No deviations were found in tendencies of changes for the two samples. The number of replicates is not relevant because the aim was to find whether the artificial light sources can imitate the effect of sunlight. The measurement of 5 independent points is enough to represent the colour data of a wood sample (Németh 1981).

One natural and two artificial irradiation types were chosen to be able to compare the effect of sunlight to the effect of the regularly applied artificial light sources. The natural sunlight irradiation was carried out between 5th of May and 19th of August, 2003 (air temp. varied 16-41 °C, max. r.h. 80% and the daily average of total solar power density was between 436-459 $W/m^2$) in Takayama (Gifu Prefecture, Japan). Geographical data for Takayama are: 36 degrees 9.3 minutes latitude and the altitude was 560 meters. The samples were exposed outside only on sunny days to determine the effect of the sunlight alone. After exposure the samples were stored in total darkness in the laboratory. The other series of specimens were irradiated with a soft irradiation source, a xenon lamp at 180 $W/m^2$, in the range of 300–400 nm, at 63 °C (black panel) and 50% r.h., in a commercial chamber (SX-75: Suga Test Instruments Co. Ltd., Tokyo). There was a quartz glass filter around the lamp. Unfortunately, the xenon lamp emits less UV radiation than sunlight. That is why a strong UV light emitter, a mercury vapour lamp also was used to irradiate specimens (HAL 800NL, installed into a KBP.659 Nippon Denchi Co. Ltd. chamber). The total light emission of the mercury lamp was 320 W, and the samples were located 64 cm from the lamp. The air temperature in the chamber was 26 °C. The emission spectrum of the mercury lamp contains 31% UV-A (380–315 nm), 24% UV-B (315–280 nm) and 25% UV-C (>280 nm) radiation. The total irradiation time was 200 hours for sunlight and the xenon light, and 20 hours for the mercury light. The irradiation with a mercury lamp produces much greater changes than the other two types of irradiation. That is why the irradiation was stopped after 20 hours.

The colour of the wood specimens was measured before and after irradiation. The exposures were interrupted after 5; 10; 20; 30; 60 and 120 hours (for the mercury lamp these data were ten times smaller) to measure the colour data. The colour measurements were carried out with a colorimeter (SE-2000 Nippon Denshoku Industries Co. Ltd., Tokyo). The *L**, *a**, *b** colour co-ordinates were calculated based on the D65 light source. The infrared (IR) spectra measurements were made with a JASCO FTIR double beam spectrometer equipped with a diffuse reflectance unit (JASCO: DR-81). The resolution was 4 $cm^{-1}$ and 64 scans were obtained and averaged. The background spectrum was obtained against an aluminium plate. The spectral intensities were calculated in Kubelka-Munk (K-M) units. The spectra were normalised to the band between 1350 $cm^{-1}$ and 1405 $cm^{-1}$, and a two point baseline correction at 3800 $cm^{-1}$ and 1900 $cm^{-1}$ was carried out.

## 3 RESULT AND DISCUSSION

The colour of wood is mainly determined by its extractives. Thus the colour change originates mostly in the chemical changes of the extractives. The degradation of the main components (cellulose, hemicelluloses and lignin) of wood can be monitored by IR spectroscopy. However the chemical changes of extractives cannot be followed by IR spectroscopy because they are present in a relatively low concentration. The IR spectrum of wood is rather complex because of its complicated chemical structure. The calculation of the difference spectrum (the spectrum after irradiation minus the bulk spectrum) is often used for presenting the changes. In this case only those bands are visible which showed alterations. *Figure 1* presents the difference IR spectrum "fingerprint region" of black locust created by 200 hours of irradiation by sunlight. After irradiation, the carbonyl band between 1680 and 1850 $cm^{-1}$ increased and the peak of the aromatic skeletal vibration originating from lignin at 1596 and 1510 $cm^{-1}$ decreased together with the guaiacyl vibrations at 1275 $cm^{-1}$ as noted in previous studies (Horn et al. 1994, Pandey – Theagarayan 1997, Kosikova – Tolvaj 1998, Ohkoshi 2002, Müller et al. 2003, Sudiyani et al. 2003, Mitsui – Tsuchikawa 2005). Usually, two peaks develop in the 1680–1850 $cm^{-1}$ region during the exposure of wood to UV radiation. It was strengthened by 2D IR correlation spectroscopy (Popescu et al. 2011). The splitting of the ether bands is also visible at 1171 and 1138 $cm^{-1}$ wavenumber.

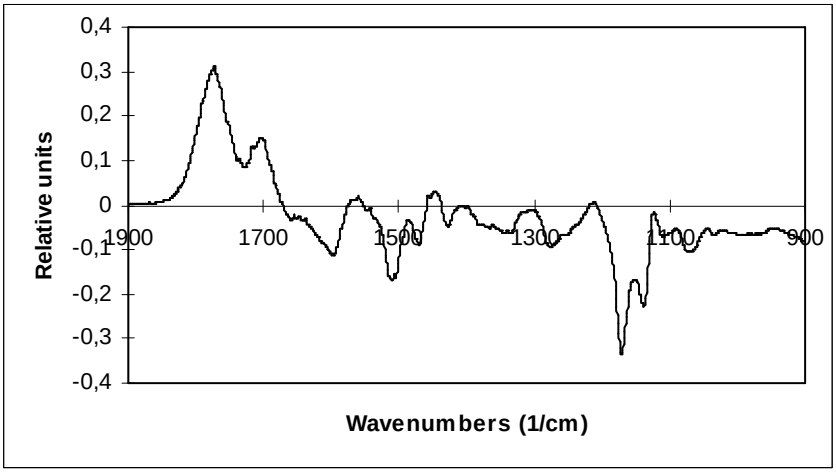



*Figure 1. Difference IR spectrum of black locust after 200 hours irradiation by sunlight*

The degradation of lignin is demonstrated by the decrease of IR bands at 1596 and 1510 $cm^{-1}$. This is presented in *Figure 2*. These data were created by sunlight irradiation.

The trend lines in *Figure 2* are parallel. The only difference is that lignin molecules of black locust which absorb at 1510 $cm^{-1}$ did not undergo changes during the first 10 hours of irradiation. This phenomenon shows the protecting effect of extractives at the beginning of the irradiation. This period was shorter than 10 hours for the artificial light sources. If the extractives have already degraded, the protection of lignin breaks down. This is presented by the parallel trend lines. The lignin of unprotected poplar suffered the greatest degradation in all cases. The greatest absorption decrease of poplar was 0.34 units at 1510 $cm^{-1}$. It was generated by a mercury lamp during 20 hours of irradiation. The decrease of the absorption at 1596 $cm^{-1}$ was similar but smaller than the decrease of the absorption at 1510 $cm^{-1}$.

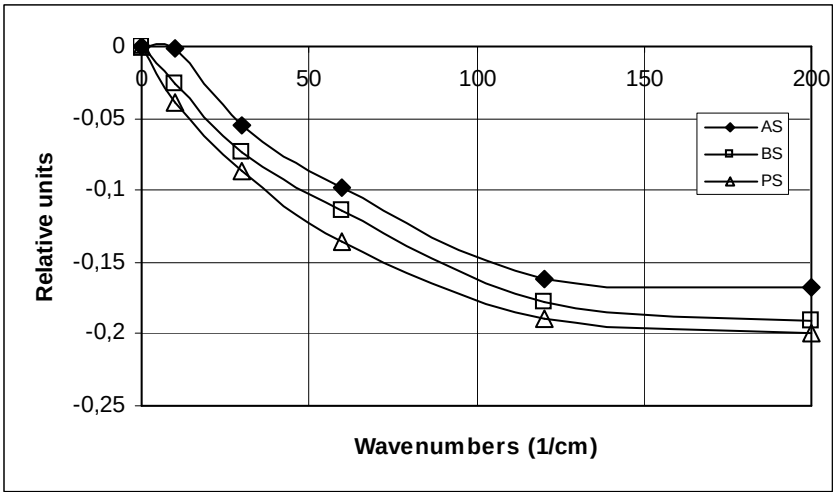



*Figure 2. IR absorption change of aromatic ring for lignin at 1510 $cm^{-1}$ produced by sunlight irradiation (for abbreviations see Fig. 7)*

The natural colour of wood is beautiful but at the same time it is extremely sensitive to light radiation. UV light in particular causes remarkable colour changes. Colour change is more pronounced initially during exposure than the other indicators. In some species, this colour change is notable and visible to the unaided eye after only a few hours of radiation. During the first 20 hours of light irradiation by sun or a xenon lamp, the lightness change was rapid as shown in *Figure 3* and *Figure 4*. The rapid period of bleaching caused by sunlight contributed 76% to the total change in black locust, 58% in beech and 39% in poplar. The extraordinary behaviour of black locust can be explained by its high extractive content. The UV light degrades the extractives followed by the rapid oxidation of the degradation products. This is presented by the rapid lightness decrease. The modified chromophores act as a kind of energy trap which slow down the photodegradation of the main wood components (Németh et al. 1992).

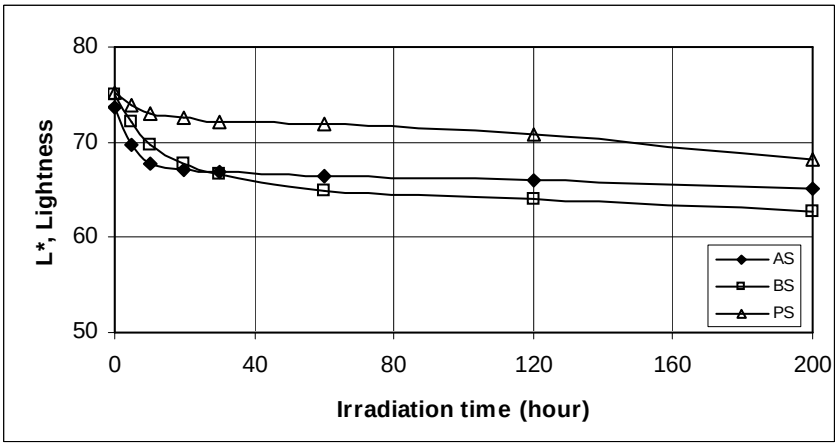



*Figure 3. Lightness change of black locust (A), beech (B) and poplar (P) samples caused by sunlight (S)*

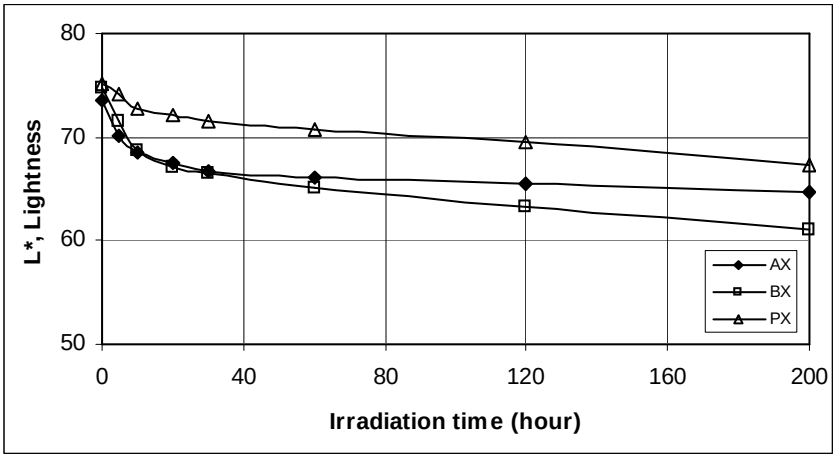



*Figure 4. Lightness change of black locust (A), beech (B) and poplar (P) samples caused by a xenon lamp (X)*

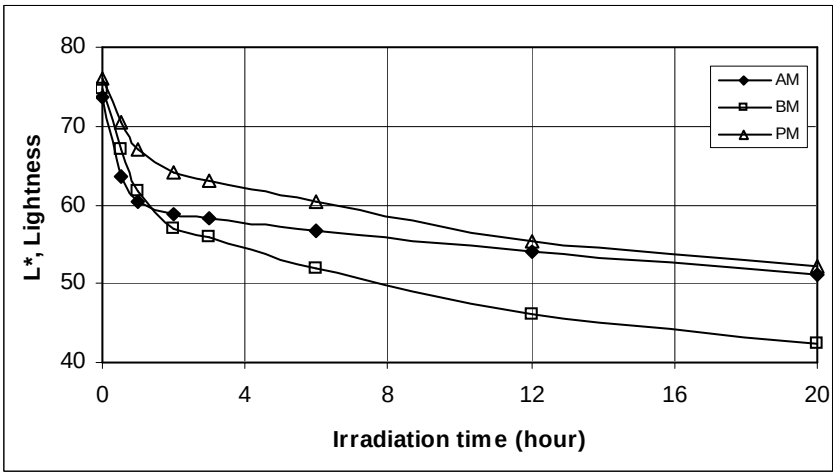



*Figure 5. Lightness change of black locust (A), beech (B) and poplar (P) samples caused by a mercury lamp (M)*

This effect is thought to be the main reason for the slower lightness change of black locust after 20 hours of irradiation compared to other examined species. It is well visible in *Figures 3–5*. The xenon lamp created a slightly greater change than sunlight. The effect of the mercury lamp irradiation (*Figure 5*) differs from that caused by a xenon lamp or natural exposure. The decrease of lightness was intensive in the whole examined time period. The mercury light irradiation was applied for up to 20 hours. This is 10 times shorter than for the other two light sources. Considering this period, the tendencies of changes were similar as it is visible comparing *Figures 3*, *4* and *5*. The mercury lamp caused more intensive colour changes than the other two light sources because the emission spectrum of the mercury lamp is mainly in the UV region (80%).

Comparing the decreasing tendency of lightness caused by the three light sources (*Figures 3–5*), we conclude that the behaviour of black locust is different to that of the other two species irrespective of the light source. Poplar samples produced the smallest lightness change. The rapid decrease in the first 10% of irradiation time is missing because poplar wood hardly has any extractives. Therefore poplar is a good reference to determine the effect of extractives. Beech has high extractive content; however the main extractives of beech and black locust are chemically different. Extractives of beech provide less protection against photodegradation compared to the extractives of black locust.

The change of red hue (*a**) presents even more deviation of black locust compared to the other two wood species (*Figures 6–8*). Only the strong UV light emitting mercury lamp produced similar changes (*Figure 8*) in all cases. The red hue shift of black locust was extremely fast during the first 5 hours of treatment. This increase represents 59% of the total increase of *a** during the xenon lamp irradiation. At the same time, the red hue shift of poplar and beech was slight. In the second period of the treatment (after 60 hours), the red hue of black locust hardly changed, while that of the other two species increased continuously. The xenon lamp induced a greater red hue shift than sunlight during the first 80 hours of exposure. The tendency of the changes was highly different at the beginning of the irradiation. The most intensive change of red colour was created by the mercury lamp irradiation.

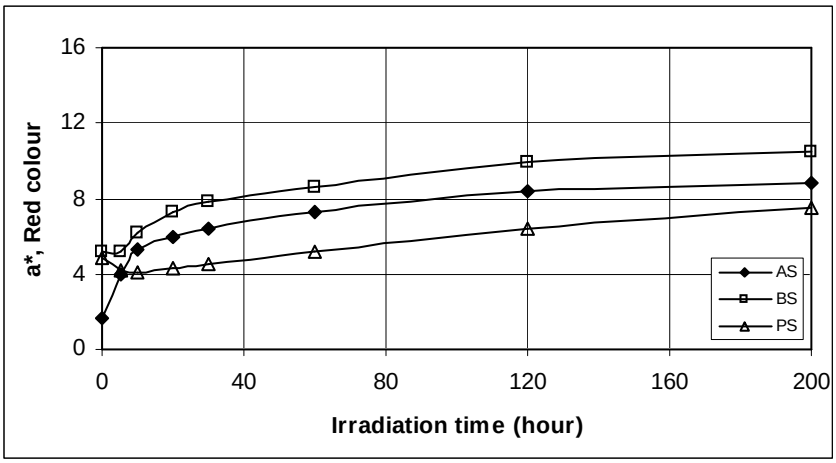



*Figure 6. Red hue shift of black locust (A), beech (B) and poplar (P) samples caused by sunlight (S)*

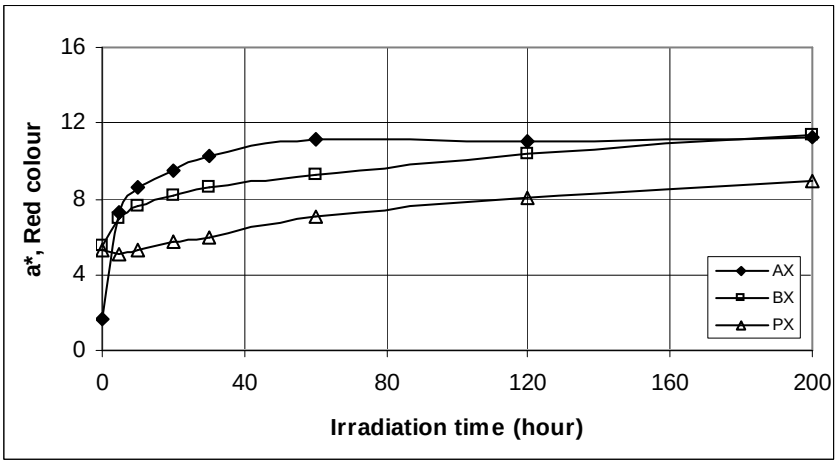



*Figure 7. Red hue shift of black locust (A), beech (B) and poplar (P) samples caused by a xenon lamp (X)*

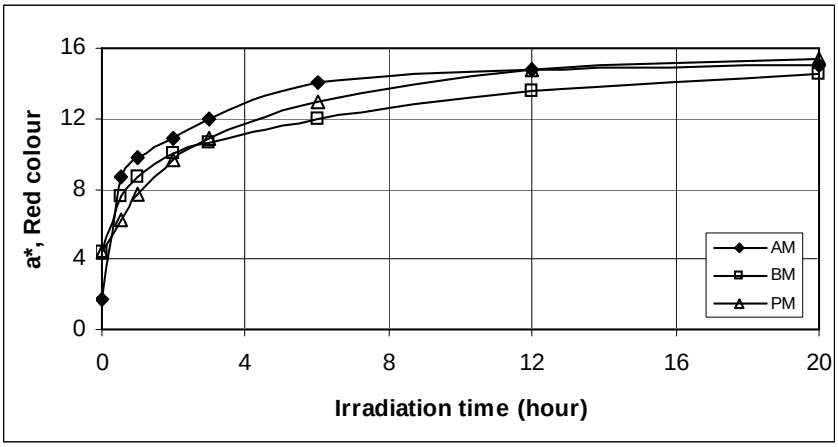



*Figure 8. Red hue shift of black locust (A), beech (B) and poplar (P) samples caused by a mercury lamp (M)*

The yellow hue of the samples increased continuously during the irradiation (*Figures 9–11*). As the initial yellow colour (*b**) of black locust is higher than that of the other two species, the trend line of black locust is located above the others. The only exception is in *Figure 11* where the low extractive content of poplar was unable to protect the lignin against the strong UV radiation of the mercury lamp. The degradation products of lignin created the increase of yellow colour. This observation is verified by the IR data (*Figure 2*). The xenon lamp induced a greater yellow shift than sunlight during the first 40 hours of exposure. The tendency of the changes was highly different at the beginning of the irradiation.

All three colour co-ordinates demonstrate that the photodegradation properties of black locust are more accentuated compared to the same properties of poplar. The extractives of black locust samples suffered rapid degradation at the beginning of the light irradiation. The degradation products partly protected the surface of the samples during further light irradiation. In contrast, poplar suffers continuous degradation during irradiation since it has no extractives to protect the surface.

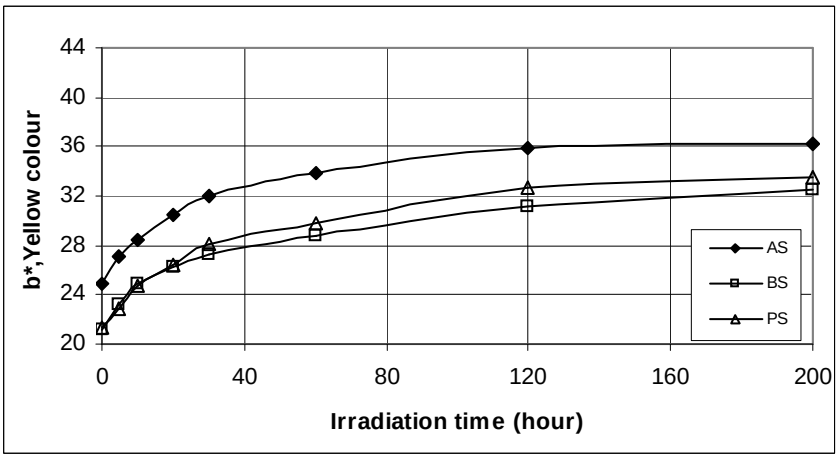



*Figure 9. Yellow hue shift of black locust (A), beech (B) and poplar (P) samples caused by sunlight (S)*

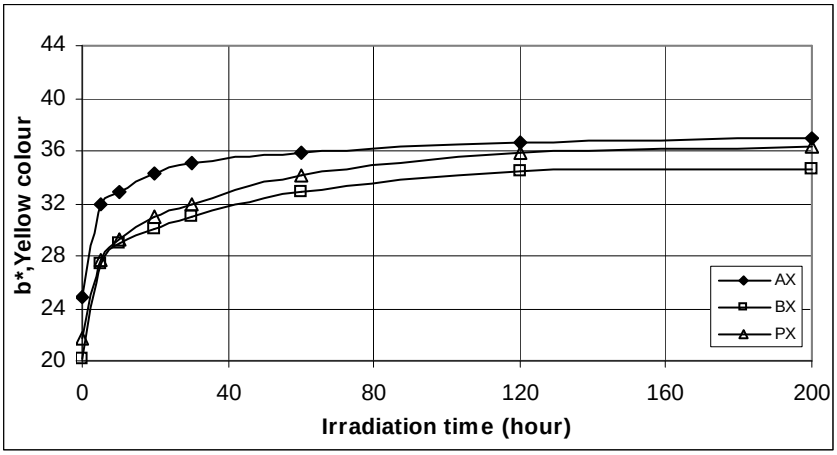



*Figure 10. Yellow hue shift of black locust (A), beech (B) and poplar (P) samples caused by a xenon lamp (X)*

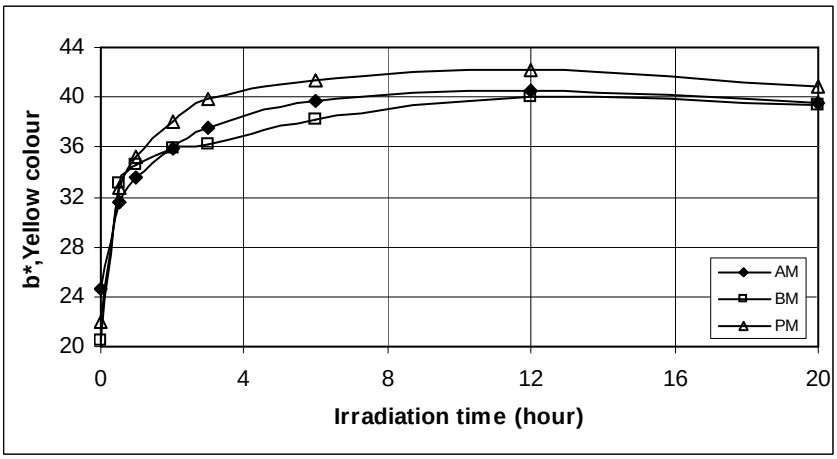



*Figure 11. Yellow hue shift of black locust (A), beech (B) and poplar (P) samples caused by a mercury lamp (M)*

## 4 CONCLUSIONS

The colour change of black locust was more intensive at the beginning of the irradiation than that of the beech and poplar. The degradation of the aromatic structure of lignin (absorbing at 1510 and 1596 $cm^{-1}$) in black locust was minor compared to the same changes of beech and poplar during the first 10 hours. These results show that the extractives of black locust are highly sensitive to light irradiation. These extractives suffered rapid degradation at the first period of the light irradiation. Lightness change, red and yellow hue shifts of black locust revealed that the degradation products protect other extractives and partly protect the lignin. This protection was confirmed by the changes of the IR spectra. The IR spectra of the investigated species showed that poplar is the most exposed because of its lack of extractives. Xenon light is able to simulate the effect of sunlight during weathering only after long exposure times. In the short term, the changes of red and yellow hues are faster and greater from xenon light irradiation compared to sunlight. The mercury lamp as a strong UV emitter is suitable to study photodegradation but unable to simulate the effect of sunlight.

## REFERENCES

HORN, B.A. – QIU, J. – OWE, N.L. – FEIST, W.C. (1994): FT-IR Studies of weathering effects in western redcedar and southern pine. Appl. Spectrosc. 48: 662–668

KAWAMURA, F. – OHASHI, H. – KAWAI, S. – TERATANI, F. – KAI, Y. (1996): Photodiscoloration of Western Hemlock (*Tsuga hterophilla*) Sapwood I. Actual conditions upon photodiscoloration of wood parts. Mokuzai Gakkaishi 42 (3): 293–300

KOSIKOVA, B. – TOLVAJ, L. (1998): Structural changes of lignin- polysaccharide complex during photodegradation of Populus grandis. Drevarski Vyskum 43: 37–46

MITSUI, K. – TAKADA, H. – SUGIYAMA, M. – HASEGAVA, R. (2001): Changes in the properties of light-irradiated wood with heat treatment. part 1. effect of treatment conditions on the change in color. Holzforschung 55 (6): 601–605

MITSUI, K. (2004): Changes in the properties of light-irradiated wood with heat treatment. Part 2. Effect of light-irradiation time and wavelength. Holz Roh Werkstoff 62: 23–30

MITSUI, K. – MURATA, A. – TSUCHIKAWA, S. – KOHARA, M. (2004): Wood photography using light irradiation and heat treatment. Color Res. Appl. 29 (4): 312–316

MITSUI, K. – TSUCHIKAWA, S. (2005): Low atmospheric temperature dependence on photodegradation of wood. J. Photochem. Photobiol. B: Biol. 81: 84–88

MITSUI, K. – TOLVAJ, L. – PAPP, G. – BOHUS, J. – SZATMÁRI, S. – BERKESI, O. (2005): Changes in the properties of light-irradiated wood with heat treatment. Part 4. Application of laser. Wood Research 50 (1): 1–8

MOLNÁR, S. – BARISKA, M. (2002): Wood species of Hungary. Szaktudás Kiadó Ház, Budapest, 54–60

MÜLLER, U. – RÄTZSCH, M. – SCHWANNINGER, M. – STEINER, M. – ZÖBL, H. (2003): Yellowing and IR-changes of spruce wood as result of UV-irradiation. J. Photochem. Photobiol. B: Biol. 69: 97–105

NÉMETH, K. (1981): Színmérés a faiparban I. A természetes fa színmeghatározása [Colour measurement in wood industry I. Colour determination for natural wood.] Faipar 31 (9): 257–261

NÉMETH, K. – VANÓ, V. – FAIX, O. (1992): The effect of wood extractives on the photodegradation of wood. EWLP Conf. Grenoble, France. 191–192

OHKOSHI, M.J. (2002): FTIR-PAS study of light-induced changes in the surface of acetylated or polyethylene glycol-impregnated wood. J. Wood Sci. 48: 394–401

OLTEAN, L. – HANSMANN, C. – NÉMETH, R. – TEISCHINGER, A. (2010): Wood surface discolouration of three hungarian hardwood species due to simulated indoor sunlight exposure. Wood Research 55(1):49–58

OTA, M. – OGATA, H. – JONO, Y. – HIROTA, K. – ABE K. (1997): Light-induced color changes of acetylated veneers of kiri (Paulowniatomentosa Steud.). Mokuzai Gakkaishi 43: 785–791

PANDEY, K.K. – THEAGARAJAN, K.S. (1997): Analysis of wood surfaces and ground wood by diffuse reflectance (DRIFT) and photoacustic (PAS) Fourier transform infrared spectroscopic techniques. Holz Roh Werkstoff 55: 383–390

PANDEY, K.K. (2005): A note on the influence of extractives on the photo-discoloration and photo-degradation of wood. Polym. Degrad. Stab. 87 (2):375–379

PANDEY, K.K. (2005): Study of the effect of photo-irradiation on the surface chemistry of wood. Polym. Degrad. Stab. 90(1): 9–20

PANDEY, K.K. – VUORINEN, T. (2008): Comparative study of photodegradation of wood by a UV laser and a xenon light source. Polym Degrad Stab 93 (12): 2138–2146

PASTORE, T.C.M. – SANTOS, K.O. – RUBIM, J.C. (2004): A spectrocolorimetric study on the effect of ultraviolet irradiation of four tropical hardwoods. Bioresource Technol. 93 (1): 37–42

PODGORSKI, L. – MERLIN, A. – DEGLISE, X. (1996): Analysis of the natural and artificial weathering of a wood coating by measurement of the glass transition temperature. Holzforschung 50: 282–287

POPESCU, C.M. – POPESCU, M.C. – VASILE, C. (2011): Structural analysis of photodegraded lime wood by means of FT-IR and 2D IR correlation spectroscopy. Int. J. Biol. Macromolecules 48: 667–675

SUDIYANI, Y. – IMAMURA, Y. – DOI, S. – YAMAUCHI, S. (2003): Infrared spectroscopic investigations of weathering effects on the surface of tropical wood. J. Wood Sci. 49: 86–92

TOLVAJ, L. – FAIX, O. (1995): Artifical ageing of wood monitored by drift spectroscopy and cie l*a*b* color measurements. i. effect of uv light. Holzforschung 49 (5): 397–404

TOLVAJ, L. – MITSUI, K. (2005): Light source dependence of the photodegradation of wood. J. Wood Sci. 51 (5): 468–473

# Technical Papers

# Comparison of Chloroplast Genomes of English Yew (*Taxus baccata* L.) and Japanese Black Pine (*Pinus thunbergii* Parl.)

Norbert FRANK[a*] – Berthold HEINZE[b]

[a] Institute of Silviculture and Forest Protection, Faculty of Forestry, University of West Hungary, Sopron, Hungary
[b] Department of Genetics, Federal Research Centre for Forests, Natural Hazards and Landscape (BFW), Vienna, Austria

**Abstract** – Previous studies based on of allozyme polymorphisms in *Taxus baccata* (English yew) have been concerned mainly with inheritance analysis and genetic differentiation within and among yew stands. Our primary goal was to find chloroplast primer restriction enzyme combinations which reveal genetic polymorphisms in the chloroplast genome of samples from Central Europe but no polymorphisms were detected. However, we were able to map location of successful primer combinations on the published chloroplast genome of Japanese black pine (*Pinus thunbergii*) and to compare the primer positions with those of two other completely sequenced conifer genomes. Yew sequences in GenBank were also placed onto our map. As we were able to amplify a number of hitherto unexplored spacers between genes, the map of chloroplast structure shows that gene order is likely to be identical between *Taxus* and *Pinus* over large parts of the chloroplast.

**similarity / chloroplast / *English yew* / genetic polymorphism**

**Kivonat – Az európai tiszafa és a japán feketefenyő cpDNS összehasonlító vizsgálata.** Korábbi tanulmányok az európai tiszafa (*Taxus baccata*) izoenzim polimorfizmus öröklődésének elemzésével, az állományokon belüli és állományok közötti genetikai differenciálódásával foglalkoztak. Vizsgálataink elsődleges célja olyan kloroplasztisz restrikciós enzimek azonosítása, amelyekkel a közép-európai minták genomjának polimorfizmusa detektálható. Sajnos e polimorfizmust nem tudtuk egyértelműen igazolni, azonban működő primerkombinációk alapján megrajzoltuk a japán feketefenyő (*Pinus thunbergii*) korábban publikált kloroplasztisz genomjára illesztett genomtérképet, amelyhez a GenBank-ban található európai tiszafa szekvenciákat is felhasználtuk. Számos, eddig még fel nem derített, nemátíródó szekvenciát tudtunk amplifikálni. Az így elkészített géntérkép azt valószínűsíti, hogy a vizsgált két faj kloroplasztiszában a gének sorrendje nagymértékben hasonló.

**hasonlóság / kloroplasztisz / Európai tiszafa/ genetikai polimorfizmus**

## 1 INTRODUCTION

English yew (*Taxus baccata* L.) has a disjunct distribution in Europe, Asia Minor, the Caucasus and North Africa, extending northward up to 61° N latitude in Scandinavia and southward to the Mediterranean, the Crimea and the Caucasus, eastward to the Baltic Sea and

[*] Corresponding author: frank@emk.nyme.hu; H-9400 SOPRON, Bajcsy Zs. u. 4.

the Carpathians and westward to England and Ireland. Its altitudinal range reaches up to 2000 m elevation in Spain and 1200 m in the Alps (Atlas Florae Europeae 2008).

In Hungary, the largest natural occurrence of yew is found in the beech forest of the Bakony Mountains, near Szentgál (Majer 1980).

Previous studies based on allozyme polymorphisms have mainly been concerned with inheritance analysis and genetic differentiation within and among stands on local or regional scale (Lewandowski et al. 1992, Thoma 1992, Hertel 1996, Rajewski – Large 1997, Rajewski et al. 1999, Rajewski et al. 2000, Collins et al. 2003, Frank 2006). The levels of polymorphism were comparable to many European forest trees.

Our primary goal was to find chloroplast primer and restriction enzyme combinations to detect genetic polymorphisms in the samples from Central Europe. Additionally, we wanted to compare the results of our amplification trials with chloroplast structures of three fully sequenced conifer chloroplasts.

## 2 METHODS

Approximately 125 trees located in the natural yew stand in (Bakony Mts.) Szentgál in Hungary (Latitude: 47°6'31.59" N, Longitude: 17°46'59.32" E), on the territory of the Balaton Upland National Park, plus another three specimens (one male, one female, and one fastigiate "Irish yew") were sampled in the arboretum at Mariabrunn of the Federal Research Centre for Forests (BFW) in Vienna. The selection of primer pairs was based on the chloroplast PCR primer database (http://bfw.ac.at/rz/bfwcms.web?dok=977, Heinze 2007), with *Pinus* as a reference (see *Table 1* for primers).

The PCR reaction mix contained 1 µL DNA, 1x PCR buffer, 2.0 mM $MgCl_2$, 200 nM dNTPs (Invitrogen), 0.2 µM of each primer, Taq Polymerase (Invitrogen) in a total volume of 15 µL. PCR amplification comprised 10 cycles pre-amplification (94°C – 70°C), and 35 cycles of amplification at annealing temperatures of 50°C or 55°C. Five µL of PCR products were separated by electrophoresis in a 1.5% agarose gel, in 0.5% TBE buffer using a 100 base pair (bp) ladder (Invitrogen) as a size standard, and visualized by UV fluorescence with ethidium bromide staining.

PCR products were digested with the following restriction enzymes: *AluI*, *BamHI*, *DraI*, *EcoRI*, *HaeIII*, *HhaI*, *HinfI*, *MspI*, *RsaI*. Ten µL of the total PCR products were digested in a total volume of 14 µL. The digested cpDNA fragments were separated on 2.5% agarose gels.

For drawing the chloroplast map in comparison to *Pinus thunbergii* (GenBank[1] Acc. No. NC_001631), we used the Circular Genome Viewer (CGView, Stothard and Wishart 2004). Fragments, that we amplified ourselves, are described as amplified in the literature, and appear in GenBank were drawn onto the map with their approximate positions.

Positions of the primer pairs that sucessfully amplified fragments in *Taxus baccata* were compared to *P. thunbergii*, *Cryptomeria japonica* (GenBank Acc. No. NC_010548) and *Keteleeria davidiana* (NC_011930) with the help of the seqmatchall procedure implemented on the mobyle@pasteur web site (http://mobyle.pasteur.fr/cgi-bin/portal.py?).

## 3 RESULTS

In common with previous studies, we were unable to detect any cpDNA polymorphisms in English yew samples from Central Europe. The map of chloroplast genome structure of yew

[1] GenBank is a genetic sequence database, an annotated collection of all publicly available DNA sequences. Available online: http://www.ncbi.nlm.nih.gov/genbank/

shows that we were able to amplify new fragments that were neither in NCBI (National Center for Biotechnology Information, available online: http://www.ncbi.nlm.nih.gov/), nor studied by other groups (*Figure 1*). For the spacers that we amplified, gene order is conserved between *Taxus* and *Pinus*. When comparing with the two other completely sequenced conifer chloroplasts, we were often unable to find sufficient match of the primers to the DNA sequences, or primer binding sites were so far apart or in such orientation so that amplification from these conifers (or from *Taxus*, if it had similarly arranged genes) seems unlikely. In the long run, sequencing of the total chloroplast of yew is necessary to confirm this observation (*Table 1*).

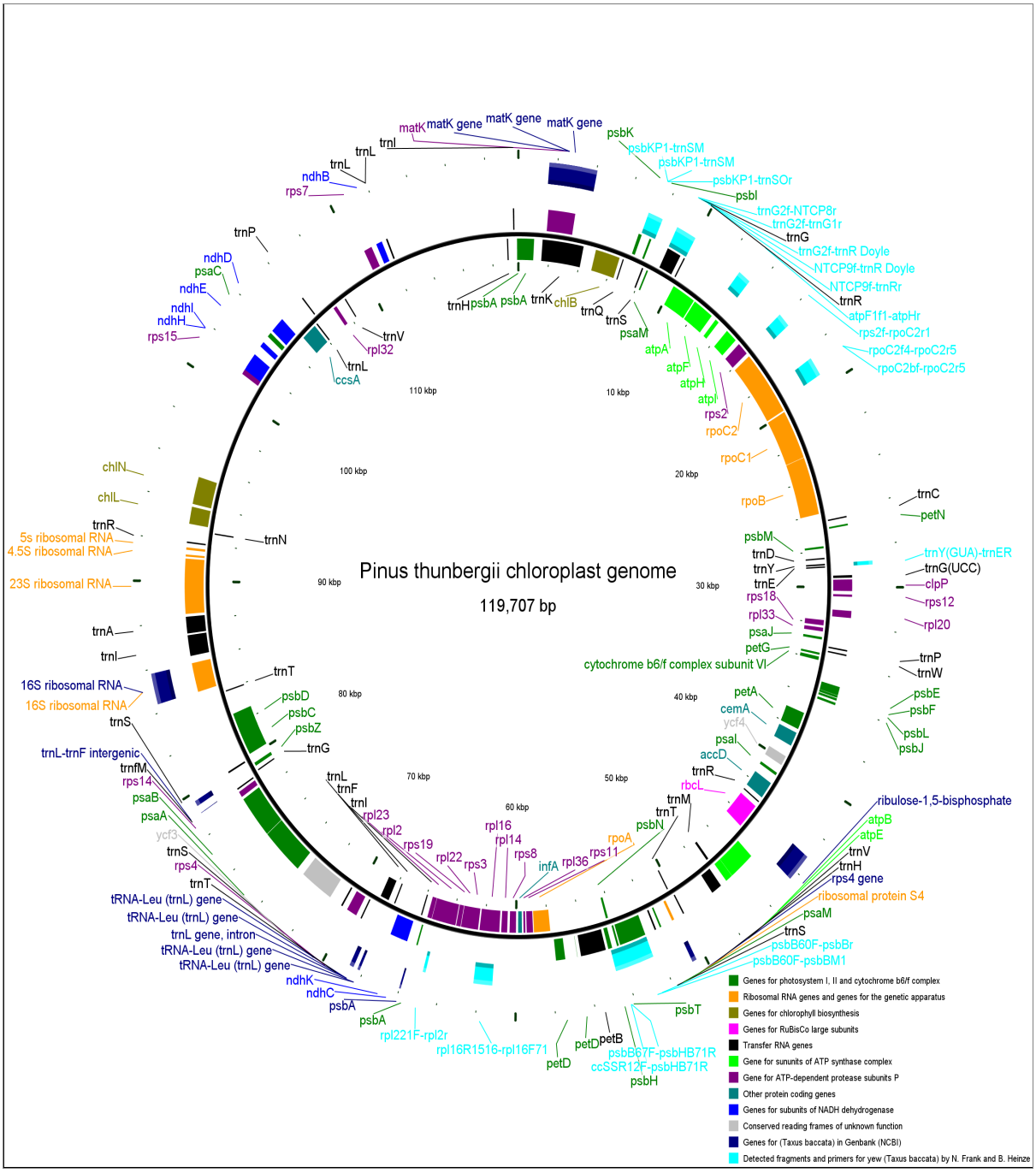



*Figure 1. Structure comparison between the chloroplast of Pinus thunbergii and fragments in the Taxus baccata chloroplast*

*Table 1: Primers selected for PCR amplification (Heinze 2007), and their binding sites in fully sequenced conifer chloroplasts*

| Primer pairs (forward-reverse) | *Pinus thunbergii* | | *Cryptomeria japonica* | | *Keteleeria davidiana* | | *Taxus baccata* |
|---|---|---|---|---|---|---|---|
| | amplification range | size (calc.) | amplification range | size (calc.) | amplification range | size (calc.) | approx. size (PCR product) |
| atpF1f1-atpHr | 12821-13516 | 695 | no match-78913 | – | 53704-no match | – | 650 |
| NTCP9f-trnR Doyle | 9616-9904 | 288 | no match-82389 | – | 50507-no match | – | 300 |
| trnG2f-trnR Doyle | 8854-9904 | 1050 | no match-82389 | – | 49685-no match | – | 1000 |
| NTCP9f-trnRr | 9616-9953 | 337 | no match-82346 | – | 50507-no match | – | 225 |
| trnG2f-NTCP8r | 8854-9547 | 693 | no match-82692 | – | 49685-no match | – | 625 |
| trnG2f-trnG1r | 8854-9635 | 781 | no match-82617 | – | 49685-no match | – | 225 |
| psbB60F-psbBr | 52424-52774 | 350 | 4013-no match | – | 91794-89390 | 2407 | 375 |
| psbB60F-psbBM1 | 52424-53262 | 838 | 4013-103343 | 99330 | 91794-no match | – | 750 |
| psbB63F-psbBB68R | 53105-53931 | 826 | 4709-no match | – | 92490-no match | – | 825 |
| psbB67F-psbHB71R | 53850-54535 | 685 | 5439-no match | – | 93220-no match | – | 750 |
| ccSSR12F-psbHB71R | 53908-54535 | 627 | no match-no match | – | 93288-no match | – | 650 |
| psbKP1-trnS0r | 7173-7980 | 807 | no match-84247 | – | 47942-91092 | 43150 | 225 |
| psbKP1-trnSM | 7173-7913 | 740 | no match-58779 | – | no match-773/91144** | – | 875 |
| rps2f-rpoC2r1 | 15781-16472 | 691 | 127423-128053 | 630 | 76110-no match* | – | 350 |
| rpoC2f4-rpoC2r5 | 18443-19366 | 923 | no match-73345 | | no match-no match | | 950 |
| rpoC2bf-rpoC2r5 | 18445-19366 | 921 | no match-no match | – | no match-no match | – | 325 |
| rpl2-21F-rpl2r | 64912-65059 | 147 | no match-19466 | – | no match-no match | – | 350 |
| rpl16R1516-rpl16F71 | 61262-62284 | 1022 | 64264-65444 | 1180 | 110807-no match | – | 600 |
| trnY(GUA)-trnE-R | 28734-28915 | 181 | 32781-62960 | 30180 | 69749-no match | – | 575 |

* no matches at the positions of the corresponding genes
** two matches

## 4 DISCUSSION

Up to now, cpDNA polymorphisms of yew were only detected in populations from the Mediterranean region (S. Gonzalez–Martinez and G.G. Vendramin, personal communication). This is in contrast to allozyme and RAPD investigations across Europe. Like many other forest tree species, *Taxus* shows typical high genetic variation within stands. Interestingly, English yew is in this respect similar to e.g. European beech (*Fagus sylvatica*), one of its strongest competitors, dominating many yew plant communities (Majer 1980, Magri et al. 2006). The absence of cpDNA polymorphism might be the consequence of a bottleneck effect. A genetic bottleneck in the past may support the view of *Taxus baccata* as a living fossil that is on the brink of extinction. The timing of such a bottleneck, however, remains highly speculative, as fossil pollen data have shown that this species was highly abundant in Central Europe in the last interglacial (Krupinski 2000, Adams 2002). Gao et al. (2007) have reported high levels of cpDNA variation in *Taxus wallichiana* in Asia; however, in a small study with material from one of these Chinese provinces, we were unable to detect variation, nor to detect differences with our Central European samples (Yafeng Wen and B. Heinze, unpublished).

Our comparison of positions of primers and amplified fragments shows that *P. thunbergii* is the best model for the *Taxus* chloroplast, among the three tested. While the taxonomic position of *Taxus* is still debated, our preliminary analysis shows that there are possibly not many gene order rearrangements in comparison to *Pinus*.

## REFERENCES

ADAMS, J.M. (compiler; 2002): Europe during the last 150,000 years (web page; URL: http://www.esd.ornl.gov/projects/qen/nercEUROPE.html). In: ADAMS, J.M. – FAURE, H. (eds.): QEN members - Review and Atlas of Palaeovegetation: Preliminary land ecosystem maps of the world since the Last Glacial Maximum. Oak Ridge National Laboratory, TN, USA. http://www.esd.ornl.gov/projects/qen/adams1.html. Accessed Sept 8, 2003.

ATLAS FLORAE EUROPAEAE (2008): Atlas Florae Europaeae – Botanical Museum – FMNH - Mozilla Firefox. Available at: http://www.fmnh.helsinki.fi/english/botany/afe/. Last accessed April 28, 2009.

CAO, C-P. – LEINEMANN, L. – ZIEHE, M. – FINKELDEY, R. (2003): Study of genetic variation and diferrentiation of Yew (*Taxus baccata* L.) stands using isozyme and DNA markers [Untersuchungen zur genetischen Variation und Differenzierung von Eibenbeständen mit Hilfe von Isoenzym- und DNS-Markern. Allgemeine] Forst-und Jagdzeitung 175: 21–28

COLLINS, D – MILL R.R. – MÖLLER M. (2003): Species separation of *Taxus baccata*, *T. canadensis* and *T. cuspidata* (Taxaceae) and origins of their reputed hybrids inferred from RAPD and cpDNA data. Am J Bot 90: 175–182

FRANK, N. (2006): A szentgáli tiszafás populáció-genetikai elemzése RAPD markerrel. [Genetic analysis with RAPD marker in the yew (*Taxus baccata* L.) population of Szentgál.] MSc. Dissertation, Szent István University, Gödöllő, Hungary. (in Hungarian)

GAO, L.M. – MOLLER, M. – ZHANG, X.M. – HOLLINGSWORTH, M.L. – LIU, J. – MILL, R.R. – GIBBY, M. – LI, D.Z. (2007): High variation and strong phylogeographic pattern among cpDNA haplotypes in Taxus wallichiana (*Taxaceae*) in China and North Vietnam. Molecular Ecology: Molecular Ecology 16(22), 4684–4698.

HERTEL, H. (1996): Inheritance of isozyme markers in English yew (*Taxus baccata* L.) [Vererbung von Isoenzymgenmarkern bei Eibe (*Taxus baccata* L.)] Silvae Genet 45: 284–290. (in German)

HEINZE, B. (2007) A database of PCR primers for the chloroplast genomes of higher plants. Plant Methods 3: 4

KRUPINSKI, K.M. (2000): Palynostratigraphic correlation of deposits in the Mazovian interglacial in Poland. Memoirs of the Polish Geological Institute (Available online at: http://www.pgi.waw.pl/english/memoirs/pr169_en.php) vol. 169.

LEWANDOWSKI, A. – BURCZYK, J. – MEJNARTOWICZ, L. (1992): Inheritance and linkage in some allozyme in *Taxus baccata* L. Silvae Genetica 41: 342–347

MAJER, A. (1980): A Bakony tiszafása [Yew of the Bakony Mountains]. Akadémiai Kiadó, Budapest. (in Hungarian)

MAGRI, D. – VENDRAMIN, G.G. – COMPS, B. – DUPANLOUP, I. – GEBUREK, T. – GOMORY, D. – LATALOWA, M. – LITT, T. – PAULE, L. – ROURE, JM. – TANTAU, I. – VAN DER KNAAP, W.O. – PETIT, R.J. – DE BEAULIEU, J.L. (2006): A new scenario for the quaternary history of European beech populations: palaeobotanical evidence and genetic consequences. New Phytologist 171: 199–221.

OTTO, K. – BRADSHAW, T. – WATSON, B. – STEWART, S. (1994): Mini DNA preparation from *Populus*. Manuscript, Dept. of Biochemistry, University of Washington, Seattle, 2 p

RAJEWSKI, M. – LANGE, S. (1997): Genetic structure in the different ontogenetic stage [Genetische Strukturen in verschieden ontogenetischen Stadien der Eibe (*Taxus baccata* L.)] Forstwissenschaftliche Diplomarbeit University of Göttingen. (in German)

RAJEWSKI, M. – LANGE, S. – HATTEMER, H.H. (1999): Genetic inference on the embryo of yew (Taxus baccata L.). Forest Genetics 6: 55–58

RAJEWSKI, M. – LANGE, S. – HATTEMER H.H. (2000): Problem of reproduction in the genetic conservation of the rare species: the example of common yew (*Taxus baccata* L.) [Reproduktion bei der Generhaltung seltener Baumarten. – Das Beispiel der Eibe (*Taxus baccata* L.).] Forest Snow Landscape Research 75: 251–266. (in German)

STOTHARD, P. – WISHART, D.S. (2005): Circular genome visualization and exploration using CGView. Bioinformatics 21: 537–539

THOMA, S, (1992): Genetische Unterschiede zwischen vier Reliktbeständen der Eibe (*Taxus baccata* L.). Forst und Holz 50: 19–24.

# Target Diameter Models for Leuce Poplar Stands Growing on Sandy Soils

Károly RÉDEI* – Zsolt KESERŰ

Hungarian Forest Research Institute, Sárvár, Hungary

**Abstract –** The fact that certain ecological factors fundamentally influencing tree growth have become unfavourable in Hungary in recent years, has led to the more extensive use of white poplar (and its hybrids) in afforestation and forest regeneration. An intensive integrated research and development work has been carried out on the growth of Leuce poplars on sandy soils, including primarily the white poplar (*Populus alba*) and its natural hybrid the grey poplar (*Populus x canescens*). The research revealed several factors influencing stand growth. The study presents a new, simplified tending operation model for Leuce poplar stands, as well as age, growing space and target diameter models suitable for qualitaty log production and for mass assortments. The simplicity of these practice-oriented models may foster the qualitative development of Leuce poplar management in Hungary.

**Leuce poplars / tending operation models / growing space regulation**

**Kivonat – Homoki Leuce-nyár állományok célátmérő modelljei.** A homoki Leuce-nyárakkal, döntően a fehér nyárral (*Populus alba*) és természetes hibridjével, a szürke nyárral (*Populus x canescens*) kapcsolatos kutatómunka intenzívebbé tételét több tényező indokolja. A fatermesztést alapvetően befolyásoló ökológiai tényezők egy részének kedvezőtlenebbé válása előtérbe helyezte e fafaj (és hibridjeinek) egyre kiterjedtebb alkalmazását az erdőtelepítések, illetve erdőfelújítások során. A tanulmány a nemzetközi szakirodalomban is hézagpótlónak tekinthető módon közli a fehérnyárasok új, egyszerűsített erdőnevelési modelljét, illetve a minőségi rönktermesztésre, valamint a tömeg-választékok előállítására alkalmas fehérnyárasok kor-növőtér-célátmérő modelljeit. A közölt gyakorlatorientált modellek egyszerűségüknél fogva nagyban segíthetik a Leuce- nyár termesztés minőségi fejlesztését Magyarországon.

**Leuce-nyárak / erdőnevelési modellek / növőtér szabályozás**

## 1 INTRODUCTION

Leuce poplars, primarily white poplar (*Populus alba*) and its most important natural hybrid the grey poplar (*Populus* x *canescens*), are tree species native to Hungary (Kopecky 1962, Szodfridt – Palotás 1973, Kopecky 1978, Rédei 1991). Due to their favourable silvicultural and growth characteristics, as well as the possibilities for the utilisation of their wood, the area they occupy is increasing continuously. The most important task facing Hungarian poplar

* Corresponding author: redei.karoly@t-online.hu, H-9600 SÁRVÁR, Várkerület 30/A.

growers is improving the quality and increasing the quantity of poplar stands for wood production (Rédei 2000).

The area occupied by the two species in 2006 was 65 000 ha (3.2% of the total forest area), with a standing volume of 9.8 million $m^3$ (163 $m^3$ $ha^{-1}$) (Führer et al. 2009). Their importance will continue to increase across the large areas of marginal land not suitable for the cultivation of hybrid poplars but able to accommodate these native species (Rédei 1991, 1994, 2000).

Other species that may be used for plantation forestry in addition to these poplar species are black locust, red oak and black walnut. Common walnut plantations may also play a role, but the silvicultural significance of this tree species is negligible.

From among the above listed tree species, the models for tending operations and the tables for age-growth space-target diameter models are suitable for production of large, quality wood material as well as mass assortments produced in white and grey poplar stands.

## 2 MATERIALS AND METHODS

The models we developed are based on a yield table for white and grey poplars. It was constructed from data gathered from 50 permanent and 40 temporary forest inventory sample plots (500–1000 $m^2$). The stands sampled were located in the vicinity of N 46° 31' 10" and E 19° 26' 46". The age of the stands varied between 5 and 45 years.

During the stand surveys, the key stand characteristics were measured. Stem volume was estimated using the following volume function (Sopp – Kolozs 2000):

$$v = 10^{-8} d^2 h^1 (h/[h-1.3])^2 \left[-0.4236 d\, h + 12.43 d + 4.6 h + 3298\right] \quad (1)$$

where $v$ is stem volume ($m^3$), $d$ is diameter at breast height (cm) and $h$ is tree height (m).

The regression analyses were calculated using a computer-based statistics programme. The expected height values of the stands at the reference age (25 years) according to the yield classes are: 24.2 m, 21.6 m, 19.0 m, 16.4 m, 13.8 m and 11.2 m. On the basis of the guide curve and the reference age (100%), a percentage value was calculated at any age and for any yield class. The yield table was constructed using the following formulae and coefficients; (a detailed dataset is available from the authors):

1. Age of stand ($A$)
2. $H_m$ = average height of stand (height of dominant and co-dominant trees) in m:
   $H_m = 1.21592 \times (1\text{-}e^{-0.09236A})^{1.8334}$
3. $D_m$ = average DBH of stand in cm:
   $D_m = 1.58356 + 0.73502 \times H_m + 0.01571 \times H^2_m$
   with $R^2 = 0.886$
4. $V_m$ = volume of stand in $m^3$ $ha^{-1}$
   $V_m = BA_m \times H \times F_m$, where
   $H \times F_m$ = form-height quotient
   $H \times F_m = 1.96791 + 0.40778 \times H_m$
   with $R^2 = 0.923$
5. $BA_m$ = basal area of stand in $m^2$ $ha^{-1}$:
   $$BA_m = \frac{D_m^2 \times \pi}{4 \times 10000} \times N_m$$

6. $N_m$ = stocking density of stand in trees ha$^{-1}$:
   $N_m = e^{8.75483-0.83879 \ln Dm}$
   with $R^2$=0.826

## 3 SIMPLIFIED TENDING OPERATION MODEL FOR LEUCE POPLAR STANDS

Leuce poplars are fast-growing species. Seedlings quickly emerge from competition with weeds. In-line and inter-row weeding is required in the first years after plantation establishment by means of seeding, as is the removal of failed plants. During tending operations, it is important to take into consideration the fact that plantations consist of trees of varied genetic make-up (genotype). In terms of the demand for light, it is worth remembering that Leuce poplar reacts extremely strongly to light availability. It also tolerates shade very well (Tóth 1996, Führer et al. 2009).

*Table 1* contains a simplified tending operation model for Leuce poplar (white and grey poplar) stands. The table was compiled using data obtained from 70 long-term yield and

experimental tending operation plots. The choice of the most suitable spacing depends on the quality of the planting material and the particular site conditions. In this table, all of the data are presented in six yield classes. The mean tree height is the most important model factor because it determines the timing of the particular tending operation. Thinnings (to expand the available growing space) must be carried out when the stocking density approaches the stem number quoted in the table.

In plantations planted at spacings of either 3 x 3 m or 3 x 2 m, there is no need for thinning, except where the sapling growth is unstable. With initial spacings narrower than 3 x 1 m or 2.5 x 1 m, one or two thinnings are recommended for plantations established with white poplar clones.

*Table 1. Simplified tending operation model for Leuce poplars (white and grey popla) stands*

| Tending operation | | | | | Stocking density (stems ha$^{-1}$) | |
|---|---|---|---|---|---|---|
| | Number | To be carried out in year… | To be carried out at $H_m$ (m) and yield class… | | Tending operation | |
| | | | | | before | after |
| Cleaning | 1 | 5–10 | 6 | I–VI | >3000 | 3000 |
| | 2 | 11–14 | 8–11 | I–VI | 3000 | 1300–1800 |
| Thinning | 1 | 15–20 | 12–17 | I–V | 1300–1800 | 650–1200 |
| | 2 | 21–25 | 16–23 | I–IV | 650–1200 | 350–600 |
| Final cutting | | 40 | | I–II | | 350–400 |
| | | 30–35 | | III–IV | | 500–600 |
| | | 25–30 | | V | | 800–900 |
| | | 20–25 | | VI | | 1000–1100 |

*Remarks for the use of the tending operation model:*

- When planning the thinning operation, the better the estimated yield class, the lower the stem number value after thinning to be applied.
- White poplar stands in yield classes V–VI are not suitable for quality wood production.

## 4 AGE, GROWING SPACE AND TARGET DIAMETER MODELS FOR LEUCE POPLAR STANDS

In plantation forestry, *the timing of the expansion of the available growing space* is significant for reaching the target assortments by maintaining the near optimal stocking density per hectare (growth space). The ecological factors of the site essentially define the target assortments; for example, whether the opportunity for the production of sizeable, quality wood material (panel log, saw-log) exists or merely thinner wood assortments (cutting, pallet and box basic material), pulp, fibre, chippings and basic wooden board materials.

The data in *table 2* show that an opportunity for the production of quality, *sizeable logs is possible in white and grey poplar stands classified yield class I-III.* For white and grey poplar stands in yield class IV – assuming an average harvesting age of 30 years – a target diameter of 18 to 20 cm can be planned with great certainty. The sustainable stocking density per hectare depending on the yield class varies from 320 to 560 stems.

*Table 3* shows that stands of yield classes IV and V are suitable for the production of mass assortments, and possibly even white and grey poplar stands of yield class VI with a target diameter of 10 to 12 cm. However, the management of stands characterised by these two lowest yield classes are usually loss producing and so are unsuitable for plantation forestry.

White and grey poplar stands growing under unfavourable ecological conditions also have an earlier harvesting age (generally between 25–30 years). The sustainable stocking density varies between 620–920 stems $ha^{-1}$ depending on the yield class. In these stands, the reduction in stem number (thinning) carried out at age 15–17 does not lead to a significant increase in diameter growth.

*Table 2. Age-target diameter model for Leuce poplars (white and grey poplar) stands targeted for quality log production*

| Planned target diameter (DBH) (cm) | Yield class | Years required to reach target diameter | Stocking density (stems $ha^{-1}$) |
|---|---|---|---|
| 18 | I | 14 | 560±5% |
| 18 | II | 17 | |
| 18 | III | 21 | |
| 18 | IV | 28 | |
| 20 | I | 16 | 515±5% |
| 20 | II | 18 | |
| 20 | III | 23 | |
| 20 | IV | 32 | |
| 25 | I | 21 | 425±5% |
| 25 | II | 25 | |
| 25 | III | 37 | |
| 30 | I | 28 | 365±5% |
| 30 | II | 42 | |

*Table 3. Age-target diameter model for Leuce poplars (white and grey poplar) stands targeed for the production of mass assortments*

| Planned target diameter (DBH) (cm) | Yield class | Years required to reach target diameter | Stocking density (stems $ha^{-1}$) |
|---|---|---|---|
| 10 | IV | 11 | |
| 10 | V | 14 | 920±5% |
| 10 | VI | 19 | |
| 12 | IV | 13 | |
| 12 | V | 17 | 790±5% |
| 12 | VI | 25 | |
| 14 | IV | 15 | |
| 14 | V | 24 | 690±5% |
| 14 | VI | – | |
| 16 | IV | 20 | |
| 16 | V | 31 | 620±5% |
| 16 | VI | – | |

## 5 CONCLUSIONS

White and grey poplar plantation management for improved growth is becoming ever more significant in lowland forestry. This fact was taken into account in the conception of this study, and fed into the novel planning tools developed to help increase the value of the material produced in Leuce poplar stands. In recent decades, growth models based on stand level data have gradually been replaced by stand growth models predicated on stem number frequencies and individual tree growth models. Nevertheless, traditional tending operation models will remain very useful tools for forest management and forest inventory. The published models can be widely used in Leuce poplar management and forest inventory, such as:

- harvest scheduling for Leuce poplar stands,
- volume estimations,
- further development of silvicultural (tending operation) models for Leuce poplar stands,
- development of guidelines for local policies promoting native species, and
- growth analysis of Leuce poplar stands.

**Acknowledgements:** Research on the improvement of Leuce poplar was supported in part by the Kiskunsági Forest and Wood Processing Company (KEFAG ZRt., Kecskemét).

## REFERENCES

FÜHRER, E. – RÉDEI, K. – TÓTH, B. (2009): Ültetvényszerű fatermesztés 1. [Plantation Forestry 1.] Agroinform kiadó, Bp. 108–125 (in Hungarian)

KOPECKY, F. (1962:). A nyárak nemesítése. [Poplar breeding] In: Keresztesi B. (ed.) A magyar nyárfatermesztés. [Poplar growing in Hungary.] Mezőgazdasági Kiadó, Budapest, 83–117. (in Hungarian)

KOPECKY, F. (1978): Keresztezéses nemesítés. [Cross-breeding] In: Keresztesi B. (ed.): A nyárak és a füzek termesztése. [Poplar and willow growing.] Mezőgazdasági Kiadó, Budapest, 47–65. (in Hungarian)

LAAR, A. – AKCA, A. (1997): Forest mensuration. Cuvillier Verlag, Göttingen.

RÉDEI, K. (1991): Entwicklungsperspektiven des Anbauses der Leuce-Pappeln in Ungarn. Erdészeti Kutatások 82–83/I: 304-312.

RÉDEI, K. (1991): A fehér (*Populus alba*) és a szürke nyár (*Populus canescens*) termesztésének fejlesztési lehetőségei Magyarországon. Erdészeti Kutatások 82–83/II: 345-352.

RÉDEI, K. (1994): Ígéretes fehér nyár (*Populus alba* L.) származások fatermése a Duna-Tisza közi homokháton. [Yield of promising *Populus alba* provenances on the Danube-Tisza region sands] Erdészeti Kutatások 84: 81–90. (in Hungarian)

RÉDEI, K. (2000): Early performance of promising white poplar (*Populus alba* L.) clones in sandy ridges between the rivers Danube and Tisza in Hungary. Forestry, Vol. 73, No. 4: 407–413.

SOPP, L. – KOLOZS, L. (2000): Volume tables. ÁESZ, Budapest, 58–64.

SZODFRIDT, I. – PALOTÁS, F. (1973): Hazai nyárak. [Native poplars]. In: DANSZKY, I. (ed.): Erdőművelés II. [Silviculture II]. Mezőgazdasági Kiadó, Budapest. (in Hungarian)

TÓTH, B. (1996): Poplar and willow growing in Hungary. Forest Research Institute, Budapest.

# Guide for Authors

Acta Silvatica et Lignaria Hungarica publishes original reports and reviews in the field of forest, wood and environmental sciences. ASLH is published in serial volumes. It is online accessible under: http://aslh.nyme.hu

Submission of an article implies that the work has not been published previously (except in the form of an abstract or as part of a published lecture or academic thesis), that it is not under consideration for publication elsewhere. Articles should be written in English. All papers will be reviewed by two independent experts.

Authors of papers accepted for publication should sign the Publishing Agreement that can be downloaded from the homepage (http://aslh.nyme.hu).

All instructions for preparation of manuscripts can be downloaded from the homepage.

The publisher provides **1** copy of the Journal and 1**0** reprints for the author.

# Contents and Abstracts of the Bulletin of Forest Science

Bulletin of Forest Science (Erdészettudományi Közlemények) is a new journal supported by the Hungarian Forest Research Institute and by the Faculty of Forestry of the University of West Hungary. The recent issue (Vol. 2, Nr. 1, 2012) contains the following papers (with page numbers in brackets), the full papers can be found and downloaded in pdf format from the journal's webpage (www.erdtudkoz.hu).

Endre SCHIBERNA, Béla LETT and István JUHÁSZ:

**Theoretical considerations of evaluating economics of continuous cover forestry** … 7–19

**Abstract** – This paper discusses the theoretical aspects of evaluating the economics of continuous cover forestry identifying continuous cover forestry as a result of selection silvicultural system. A special emphasis is being put on the comparison of cash flows occurring with different timing and scale in case of selection forest and rotation forest; their effects on the economic conditions of forestry is also investigated. The analysis is using a numeric silvicultural model of a beech stand of medium productivity and quality.

Comparison is based on the annuity of cash flows of rotation forest and selection forest. At a discount rate of 0.75% annuities are on the same level, while at discount rate of 3.5% annuity of selection forest is four times larger than that of the annuity of rotation forest. However, selection system cannot be regarded as being unambiguously economically preferable, as its advantages resulted from the frequent and stable cash flows can be balanced by the same cash flows from rotation system with a suitable great area and an even distribution of age classes.

László KOLOZS and Gábor VEPERDI:

**Determination of the volume and increment of selection and transformation forests with one-variable volume functions** … 21–34

**Abstract** – Recently the selection and transformation systems became more and more important due to the close-to-nature forest management in Hungary. The application of these systems requires a more accurate estimation of the increment by diameter classes. In order to improve the accuracy and the speed of the inventory work the authors elaborated a new volume estimation method based on the data of the Growth Monitoring Network gathered for the past 15 years.

Borbála GÁLOS, Csaba MÁTYÁS and Daniela JACOB:

**The role of afforestation in mitigating climate change** … 35–45

**Abstract** – For the 21st century warming and drying of summers in Hungary are projected to be more extreme than the hemispheric average. Climate change impact studies in the region show that recurrent droughts cause growth decline and mortality of zonal forests at their lower

(xeric) limit of distribution. Forests affect the climate through their influence on surface energy fluxes and on water cycle that alter the climate change signal. Biogeophysical feedbacks of forest cover changes on the climate have been investigated for two forest cover scenarios in the 21st century, using the regional climate model REMO. For 2021-2025 the model has been driven by the potential afforestation concept (Járó and Führer 2005) assuming 7% increase of forest cover in country mean. In the same time period as well as at the end of the 21st century (2071-2100), effects of maximal afforestation (forests over all vegetated area) have been studied.

The potential afforestation has no significant influence on the climate of Hungary. For the maximal afforestation case study (2071-2100), the expected tendency of drying may be mitigated. The largest increase of precipitation has been projected to the northeastern region, where 50% of the climate change signal can be compensated, if there is enough available water in the soil. Regarding to surface temperature, the evaporative cooling effect of forests seems to dominate.

Analysing the results it has to be taken into account that in the simulations forest cover change was performed on a limited area, and the effects appear partly in adjacent regions. Also, because of the uncertainty of certain temperate forest cover parameters, results of future field measurements are needed to improve model precision, especially at higher spatial resolution.

Gábor KOVÁCS, Gábor ILLÉS, Diána MÉSZÁROS, Orsolya SZABÓ, Andrea VIGH and Bálint HEIL:

**Evaluation of changes of site parameters in the Noszlop forest district** … 47–60

**Abstract** – A new forest management planning period is near for the forest stands of the 400 ha sized Noszlop forest district. During planning future forest stands we have to take into account the ongoing changes in site conditions. Speeding changes are forcing us to develop current site evaluation methods. In our study field observations were evaluated using advanced GIS tools.

The assessment of climate data for the past 50 years revealed that the amount of precipitation has dropped by 20% while the average temperature has increased by 1.2 °C within the study area. On the basis of newly surveyed soil profiles and core samples we derived more detailed soil and site maps than it was currently available in forest management plans. By the application of digital elevation model and connected spatial datasets the site characteristics can be visualised in a more detailed extent and their changes can be assessed. It makes us capable to describe the projected future conditions for areas of interest, spatial pattern of forest climate categories and forest site parameters can be refined within the forest management data.

Zsolt KESERŰ and Károly RÉDEI:

**Tending operation models for leuce-poplars under sandy soil conditions** … 61–71

**Abstract** – The role of the Leuce-poplars, first of all of the white poplar (Populus alba) and its natural hybrid, the grey poplar (Populus x canescens) is increasing continuously in new afforestations and artificial regenerations mainly on calcareous sandy sites in the Danube-Tisza region. The study presents a simplified tending operation model as well as age – target diameter models for Leuce-poplar stands which are suitable for qualitative log production or for the production of mass sortiments. The published practice-oriented models may help to improve the quality of Leuce - poplars growing technology in Hungary.

Roland József PESZLEN and Gábor VEPERDI:

**Modification of the silver lime yield table** … 73–80

**Abstract** – Approximately 1.2-1.9% of the Hungarian forests is covered by lime stands, almost half of these forests consists of silver lime (Tilia tomentosa Mönch.). First lime tree yield table was published by Gábor Hajdu as previously lime stand growth data had not been available. During the preparation of the graphical yield tables (called nomograms) there were some indications that the original yield data had not been realistic. Within this study possible explanations were produced and suggestions are given to modify the values of volume data.

Károly RÉDEI, Imre CSIHA, Ágnes KAMANDINÉ VÉGH and János RÁSÓ:

**The effect of intermediate cuttings on the yield and value changes in black locust (Robinia pseudoacacia L.) stands** … 81–88

**Abstract** – The investigation of the effects of intermediate cuttings on the growth of stands and theirs yield change is based on maintaining of long term experimental plots and theirs systematic estimates. On the basis of the majority of the international and inland literature the total yield of stands can not be increased by intermediate cuttings in general. From the reported case studies we can draw as a conclusion, that the investigated black locust stands' total yield can not be increased by intermediate cuttings but theirs stand-quality index can be increased by 11-24% too.

Balázs KISFALUDI, Péter PRIMUSZ, József PÉTERFALVI and Gergely MARKÓ:

**Subjective condition surveying and rating of forest roads** … 89–105

**Abstract** – Maintenance is required on forest roads after their construction. To maintain the good serviceability on the road network, its condition and the expected scale of heavy traffic must be known. In view of these two parameters, maintenance works must be and can be) done in proper time and way. Using informatics and digital technology, an efficient tool can be developed, that allows fast assessment of forest road network condition. The digital subjective condition surveying and assessment system developed in our institute makes possible the logging and evaluation of the condition of 20–25 km forest roads daily. If a forestry company possesses a digital road inventory, the condition of its road network can be displayed in a GIS. By combining the condition and expected traffic data, the necessary maintenance treatments, and their expenses can be estimated. The system has been tested on more than 1000 kilometres of forest roads. On the basis of this database, deductions can be done regarding the condition of the whole cca. 3000 kilometres of Hungarian paved forest road network.

Gergely MARKÓ, Péter PRIMUSZ and József PÉTERFALVI:

**Measuring bearing capacity of forest roads with the Advanced Benkelman Beam Apparatus** … 107–121

**Abstract** – Forest roads covered with asphalt pavement represent the basis of the forest opening up networks in Hungary. If properly maintained, asphalt pavements offer a high level of service. While traffic load of forest road networks have grown, expenses on their maintenance remained lower then required in the last three decades. As a result, this roads are in poor condition, generally. Renovation projects demand the knowledge of the roads' bearing capacity. Bearing capacity measurements of roads traditionally were carried out using the

Benkelmann beam. The Benkelman beam measurements provide the maximum vertical deflection of the pavement under 50 kN of wheel load. Nowadays the bearing capacity of public roads are measured with falling weight deflectometers. Falling weight deflectometer measurements provide the full deflection basin. It is convenient to use these high precision instruments on forest roads, but their application is inefficient and costly. Researchers of the Department of Forest Opening Up developed a new method to measure the full deflection basin with the Benkelman beam. Besides the instrument improvement the authors developed a new method for the processing of the deflection basin data. New results are presented via the case study of a 2nd class opening up forest road.

Tamás MAJOR, Katalin SZAKÁLOSNÉ MÁTYÁS and Attila László HORVÁTH:

**Determination of the mechanization-affecting soil resistance at forested sites using the „3T System" electronic soil layer indicator** … 123–134

**Abstract** – Recently the selection and transformation systems became more and more important due to the close-to-nature forest management in Hungary. The application of these systems requires a more accurate estimation of the increment by diameter classes. In order to improve the accuracy and the speed of the inventory work the authors elaborated a new volume estimation method based on the data of the Growth Monitoring Network gathered for the past 15 years.

Katalin SZAKÁLOSNÉ MÁTYÁS, Attila László HORVÁTH and Tamás MAJOR:

**Tree utilization systems in the XXI. Century** … 135–149

**Abstract** – The development and improvement of suitable soil cultivation machines is unfeasible without knowing the proper characteristics of the soils. After a detailed review of the applying scientific literature we have concluded that the physical-mechanical properties of forest soils with extensive root systems have hitherto been unidentified.

The goal of our research was to study the mechanical properties of forest soils, and to investigate the effect of the tree root system on these physical-mechanical characteristics, especially on the soil resistance. In order to reveal the interdependences we have performed several soil resistance measurements.

Using mechanical and statistical methods, we have fitted a surface to the measured data points, which can predict the changes of soil resistance as a function of the diameter of the trees and of the distance from the trees. According to our measurements it can be established that around the stems in a radius of 1-1,5 meters the higher soil resistance is caused by the presence of roots.

Norbert FRANK and Viktor TAKÁCS:

**Windbreaks and shelterbelts examination by their effect on decreasing the windspeed** … 151–162

**Abstract** – Analysis of shelterbelts gives evidence for their eligibility and their multifunctionality has positive effects on environment, landscape and habitats. Not only as a system, but its elements must be considered.

Getting back to the main purpose (wind speed reduction) of belts, the numerical classification is essential for maintaining the porosity and to justify the structure. The structure has always been changing by the environmental conditions. Not only the distribution of the network, but the trees and shrubs edges also require modification of the structure for the desired effect: reduce wind speed and creating snow dumping zone in a

proper distance. Modelling provides details of the flow system: including a number of factors, local conditions and the meteorological conditions cause differences. A few degrees difference in wind direction causes "structural change" and change in wind speed, as well as the surface. Our research aims to develop methods and metrics to objectively assist the forest belt, for a long-term maintenance and development of protection effects. We have found, if the rate of wind protected and wind effected porosity is between 1.6-5 it is a well structured windbreak. Instead of the breakthrough factor this is more simple and preciously measurable value for description. While the breakthrough factor use the wind measurements and does not examine the inner spacing, the porosity consider the changes of density.

András NÁHLIK, László DREMMEL, Gyula SÁNDOR and Tamás TARI:

**Long term effects of browsing of seedlings as examined in pole stage** … 163–172

**Abstract** – According to our hypothesis browsing of seedlings and saplings by large herbivores causes long term negative changes in quantity and quality of forest stands, trees and timber.

Data were collected in the Bükk mountains, North Hungary. Sample territory was marked out in sessile oak (Quercus petraea, Mattuschka – Lieblein) pole stand previously having been browsed on seedling's stage by large herbivores. Similarly to the sample area control territory was marked out in an unbrowsed forest stand. The number of sample and control territory in oak stands was 10-10, with a 10x10 m size. We recorded the number of trees, measured the diameter at breast height (1.3 m) of the stems, the height of the trees and the malformations of the stems which occurred lower than 2 m, such as tortuosity and fork growth. The data were evaluated by Student's t-test and Mann-Whitney U test.

1. Browsing caused slight but significant, 50 cm decrease in the height of trees and a high ratio of fork growth.
2. We concluded that despite of heavy browsing activity which lasted several years, timber quality at harvesting will not decrease.
3. At the same time a slight decrease in timber output will occur.
4. Browsing will raise costs of regeneration due to the expenses of increased number of weeding.

András KOLTAY, Tamás LAKATOS, Tímea TÓTH and Zoltán ANDRÉ:

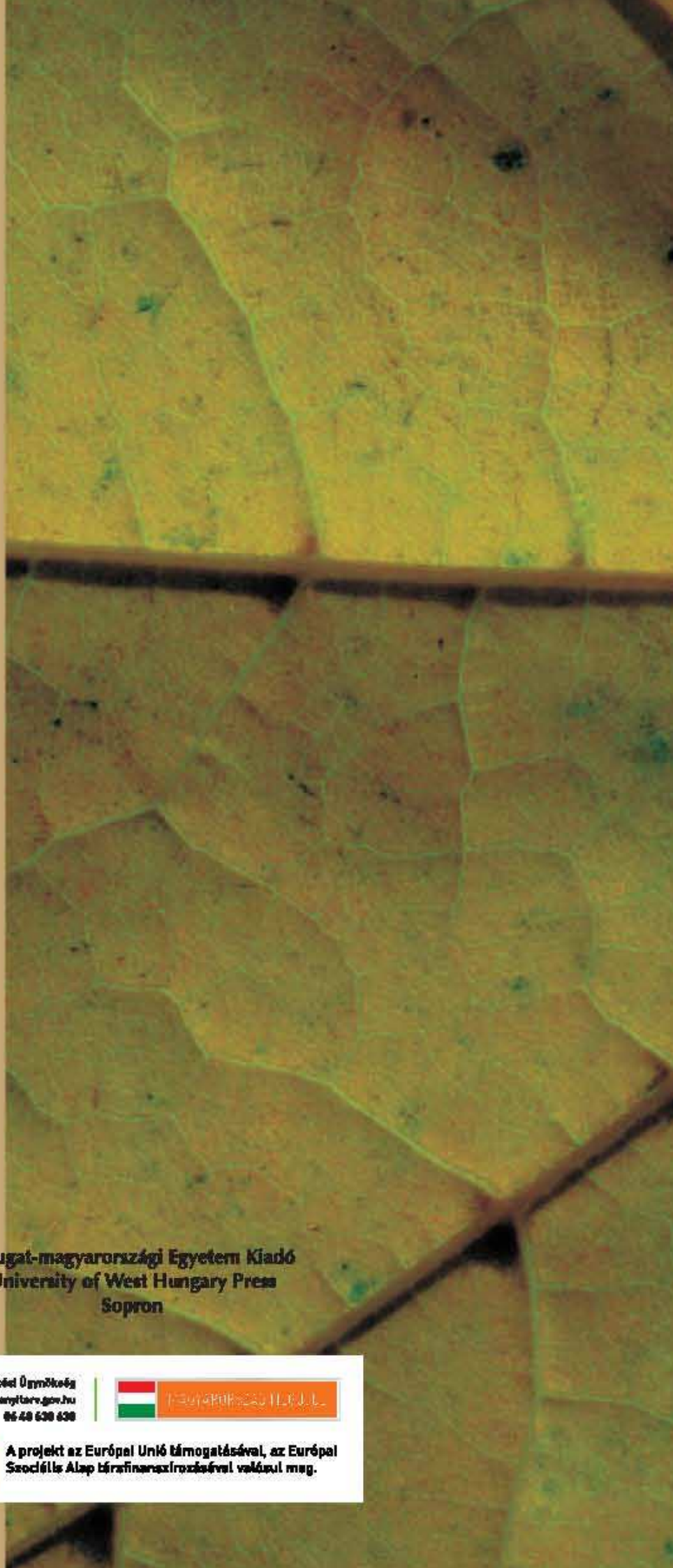
**Biological control technology against Heterobasidion annosum root rot in Scots pine stands** … 173–185

**Abstract** – In forest-ecosystems it is very important to apply selective and biological control technologies against pests and pathogens. Such method in forestry is the usage of antagonistic fungi or other micro-organisms. One of the most dangerous pathogens of conifers in Hungary is the Heterobasidion annosum root rot. Formerly Dr. Hubert Pagony applied successfully a biological control agent against it in Scots pine (Pinus sylvestris) forests. However, these research achievements fell into oblivion, although the pathogen is still present in our forests. In the past few years we re-developed this method, by the usage of Phlebiopsis gigantea, and rearranged it according to present-day requirements. We were able to produce an inoculatum, which is suitable for industrial-scale usage and manufacturing. Our experiments so far evidently claimed that this method can open new perspectives in root rot control in Scots pine and Austrian pine forests. However, the unsuccessful inoculation experiments in Norway spruce stands show that the technology in its current form is not suitable for the prevention of root rot in spruce forests.

György CSÓKA, Anikó HIRKA and Levente SZŐCS:

**Insect globalization in the Hungarian forests** … 187–198

**Abstract** – 108 alien insect species with real or potential importance for forestry have been recorded in Hungary between 1881 and 2010. The colonization rate steeply increased in the last 2-3 decades. More new species were recorded in the last 30 years (1981-2010) than in the previous 100 years (1881-1980). Order Hemiptera (particularly aphids) represent the 2/3 of the species. 46% of the species have Asian, 31% North American, 21% European, 2% African origin. 2/3 of the alien species have non native woody plants (35% conifers, 31% broadleaves) as host plant. It is quite evident that accidental introductions, spread and colonization will continue in the future. Only early recognition, increasing and disseminating the knowledge concerning them may help to slow their spread and decrease the damage caused by them.



Nyugat-magyarországi Egyetem Kiadó
University of West Hungary Press
Sopron

Nemzeti Fejlesztési Ügynökség
www.ujszechenyiterv.gov.hu
06 40 638 638

A projekt az Európai Unió támogatásával, az Európai Szociális Alap társfinanszírozásával valósul meg.